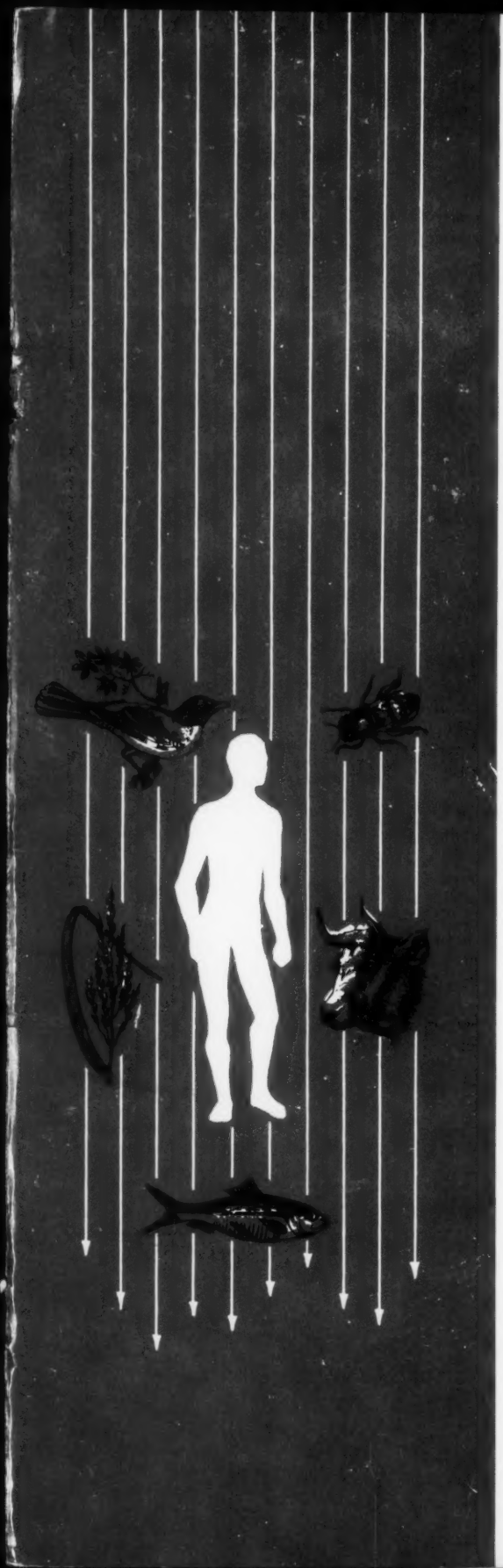

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The WORKING GROUP is comprised of representatives of the U. S. Departments of Agriculture; Commerce; Defense; the Interior; Health, Education, and Welfare; State; and Transportation; and the Environmental Protection Agency.

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Pesticide monitoring activities of the Federal Government, particularly in those agencies represented on the pesticide MONITORING PANEL which participate in operation of the national pesticides monitoring network, are expected to be the principal sources of data and interpretive articles. However, pertinent data *in summarized form*, together with interpretive discussions, are invited from both Federal and non-Federal sources, including those associated with State and community monitoring programs, universities, hospitals, and nongovernmental research institutions, both domestic and foreign. Results of studies in which monitoring data play a major or minor role or serve as support for research investigation also are welcome; however, the *Journal* is not intended as a primary medium for the publication of basic research. Manuscripts received for publication are reviewed by an Editorial Advisory Board established by the MONITORING PANEL. Authors are given the benefit of review comments prior to publication.
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EDITORIAL

These Changing Times

The well-worn phrase that "time changes everything," or as some prefer to say, "everything changes with time," has been very much in evidence recently.

First of all, you have probably noticed that there is a new "face" on this issue of the JOURNAL. The editorial staff has arranged to upgrade the quality of the cover to be more attractive and durable, yet retaining those characteristics by which the periodical is quickly recognized. Mr. Reo Duggan, Chairman of the Editorial Advisory Board of the PMJ since publication began in June 1967, has retired from service with the Food and Drug Administration. This issue of the PMJ is a fitting tribute to his untiring efforts which have brought about international recognition of the JOURNAL as one of stature in its field.

Another Monitoring Panel member, Dr. Eugene Dustman, has also retired. "Dusty" as we know him, was Director of the Patuxent Wildlife Research Center, USDI. Both Dr. Dustman and Mr. Duggan were charter members of the Panel dating back to about 1964 when it was known as the Subcommittee on Pesticide Monitoring of the Federal Committee on Pest Control. These gentlemen have provided careful and expert guidance, and their absence is a challenge to us to uphold the respected manner in which they carried out their obligations.

In May, the Working Group approved the revised Charter of the Monitoring Panel, rewritten to be compatible with the new charter for the Federal Working Group on Pest Management. Among the charges to the Panel, is to continue to have a task group serve as the interagency Editorial Advisory Board of the PMJ for publication of monitoring data. New members have been appointed. Mr. John Wessel, FDA, was chosen Chairman, and Dr. Paul Sand, USDA, Vice Chairman. The Panel pledges its full support to the Advisory Board and editorial staff.

What else has time wrought? We'll no doubt see changes in monitoring programs in keeping with monetary and personnel constraints, and changes in methods for pest control or pest management—but that's another story for another time.

Best wishes to our retirees and those who have assumed new responsibilities.

Herman Feltz
Chairman, Monitoring Panel

PESTICIDES IN PEOPLE

Total Mercury in Hair From 1,000 Idaho Residents—1971 [1]

W. W. Benson and Joe Gabica

ABSTRACT

In a study of mercury in hair from 1,000 people throughout Idaho, mercury was found in all samples. The mean concentration was 4.18 ppm, and mean levels for samples from males and females were 2.45 and 5.90 ppm, respectively. Mercury levels ranged from a low of 0.12 ppm in a male to a high of 139.0 ppm in a female. No common source of mercury exposure was found, and there is no explanation at this time for the higher mercury levels in hair samples from women.

Introduction

During the fall of 1970, a study was conducted in Idaho to determine the presence of mercury in the environment, especially in pheasants. Because of the high mercury levels found in pheasants (*1*) and subsequently in fish (*2*) and other environmental samples, the study reported here was initiated to determine mercury levels in Idaho residents.

Interest in environmental mercury as a factor in human health has increased greatly during the past several years. Reports have been published based on accidental poisonings (*3*) and poisonings resulting from occupational hazards (*4*). Most of these studies, however, have been carried out on human tissues and have not included hair samples.

The current study was designed to survey mercury levels in hair samples from persons throughout Idaho to determine if levels were higher for residents of particular areas and to determine if the consumption of fish or other sources of exposure could be correlated with mercury levels in an individual. The study was also designed to provide baseline data for future studies. Hair was studied since it was known that mercury, especially methyl mercury which is considered dangerous to human health (*5*), could be detected in hair and because such samples could be readily obtained.

Sampling Procedures

In order to compare levels in residents for different areas, sampling was carried out on the basis of the seven Idaho health districts which are apportioned by population and are approximately equal. Various organizations and individuals in each district were provided with return envelopes, printed to include name, address, age, sex, and occupation of participants as well as their consumption of fish and frequency of consumption. Initial requests met with an inadequate response; consequently, return envelopes were provided to barbers and beauticians in each district who supplied the required hair samples.

During analysis, if a sample tested above 15 ppm of mercury, an additional hair sample was requested from that individual and a blood sample was also collected. Each of these persons was also asked to complete a questionnaire listing known mercury products or activities involving mercury products and was interviewed in an effort to determine, if possible, other sources of mercury exposure. Individuals with high levels of mercury in their hair were specifically asked if they used hair coloring and conditioning preparations, diuretics, skin bleaches, and ointments; however, no queries were made concerning dental fillings.

All samples collected from men were short hair obtained close to the scalp usually from the sides and neck and, thus, represented mercury in the body at the time of sampling. In contrast, the initial samples from women were usually terminal hair ends and, depending on the

[1] From the Idaho Community Study on Pesticides, Idaho Department of Health, Statehouse, Boise, Idaho 83707.

length of hair, represented a body burden of mercury experienced previously. For example, the terminal end from hair 18 inches long, would represent that body burden experienced 12 to 18 months previously. Consequently, any repeat samples from women were requested to be hair from next to the scalp which would more accurately represent the present body burden. In all except one instance, the mercury levels were lower in subsequent samples from women. In the one exception, the level increased from 39 to 62 ppm. The current study presents only levels in initial hair samples; the subsequent samples will be correlated with blood levels of mercury in a future report.

Analytical Procedures

Each man's hair sample, since it was received as fine clippings, was mixed and an aliquot taken for analysis; a woman's hair sample, which was usually longer, had to be cut into fines and mixed before an aliquot was taken. In each case, a 1-g subsample was digested by either of two methods: Method A employed nitric and sulfuric acid digestion using the A.O.A.C. method and equipment (*6*); Method B used nitric and sulfuric acid (in a ratio of 10:1) digestion in an open 50-ml pyrex tube, placed in a boiling water bath for 2 hours. The two methods were compared by determining the final mercury levels for samples digested by both methods; and there was no significant difference (Table 1).

TABLE 1.—*Mercury levels determined after A.O.A.C. method and tube method of digestion*

Residues in ppm	
A.O.A.C.	Tube (50-ml)
1.60	1.54
2.40	3.08
0.54	0.80
21.40	21.40
86.68	100.00
2.00	1.92
1.90	2.08
4.90	5.00
1.10	1.40

The digested hair samples were then diluted to volume with distilled water in a 100-ml volumetric flask. The subsequent preparation procedure followed that outlined in the Manual of Analytical Methods (*7*) with the following modifications: The 100-ml solution was quantitatively transferred to a 300-ml BOD bottle; 2 ml of 5% potassium persulfate was added to each bottle and allowed to stand for 1 minute; 4 ml of 5% potassium permanganate was added to each bottle and allowed to stand for 1 minute; 2 ml of sodium chloride hydroxylamine sulfate was added to reduce the excess permanganate; and 5 ml of well-mixed stannous sulfate (or stannous chloride) suspension was added and the bottle immediately attached to the aeration assembly.

Prior to analyzing the hair samples, two types of equipment—the Coleman 50 mercury analyzer (C-50) and the Perkin-Elmer 303 atomic absorption spectrophotometer (P-E 303)—were compared by analyzing pheasant breast tissue. Results of analysis by each machine are given in Table 2. The C-50 provided accurate and rapid results when the mercury levels were above 0.05 μg, but for lower levels, the P-E 303 with a detection limit of 0.003 μg (*8*) was required.

Hair samples were analyzed using the C-50; however, to insure that accuracy was being maintained, the work was frequently checked by the A.O.A.C. method using the P-E 303.

If a hair sample contained residues over 15 ppm mercury, the remaining portion of hair taken from the individual was washed with a detergent having no measurable mercury, air-dried, and then reanalyzed. A

TABLE 2.—*Mercury residues in pheasant samples determined by two instruments*

Pheasant Sample Number	Mercury Residues in μG	
	P-E 303 [1]	C-50 [2]
1	.152	.06
2	.175	.10
3	.187	.18
4	.225	.18
5	.601	.81
6	.352	.37
7	.130	.10
8	.150	.10
9	.100	.10
10	.190	.10
11	.120	.07
12	.135	.10
13	.098	.09
14	.225	.19
15	.225	.16
16	.086	.05
17	.088	.07
18	.742	.70
19	.913	.82
20	.591	.59
21	.404	.47
22	.233	.27
23	.231	.21
24	.154	.29
25	.203	.28
26	.264	.32
27	.600	.42
28	.124	.23
29	1.024	1.10
30	1.300	1.22
31	.384	.42
32	.940	.81
33	.790	.77
34	.261	.14
35	.080	.07
36	.100	.07
37	.210	.15
38	.131	.10
39	.065	.07
40	.474	.53
41	.100	.10
42	.090	.07
43	.065	.06
44	.081	.05
45	.070	.07
46	.125	.22
47	.055	.14

NOTE: Coefficient of Correlation: +.97; Regressional Line of Y on X: Y = .970, X = .001; and Standard Error of Estimate of Y on X is .067.

[1] Perkin-Elmer 303 atomic absorption spectrophotometer.
[2] Coleman 50 mercury analyzer.

solvent extract of hair was not undertaken because of the solubility of many alkyl mercury compounds (*10*). Results of analyses of 12 random hair samples are given in Table 3 and indicate that there were no appreciable differences in mercury levels before and after hair washing. Any variance that did occur may have been due to the inability to make two homogeneous subsamples from the same sample. Thus, the level of mercury was not considered attributable to external contamination, but was a part of the highly proteinized hair.

After every 100 samples were analyzed, a test to determine recovery values was done by analyzing an homogenized hair sample twice, first as received and second containing a spike control with the amount of the spike being unknown to the analyst. The average recovery of spike controls was 99.7%, regardless of the digestion method used. In addition, each day a known standard was analyzed to calibrate both analytical instruments.

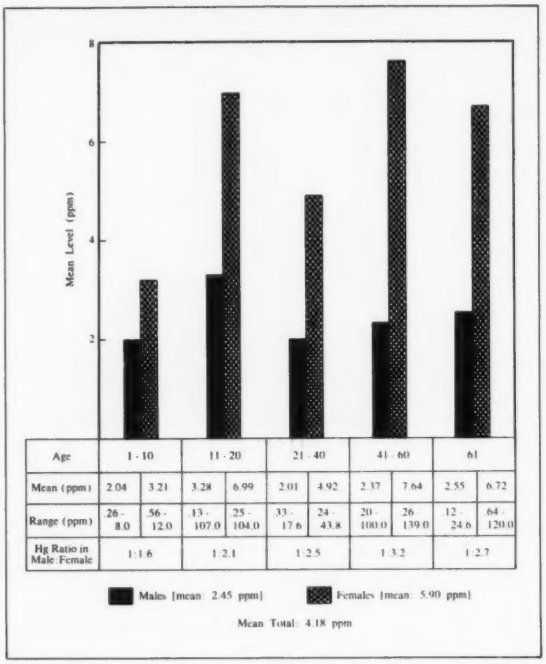
TABLE 3.—*Mercury levels in 12 random samples of hair, analyzed as received and after washing*

Mercury Residues in ppm	
Hair as Received	Washed Hair
13.86	25.60
17.40	16.40
107.00	132.00
22.40	17.60
38.00	26.40
36.00	24.00
39.00	31.00
15.20	10.00
18.40	22.00
13.60	12.00
21.20	23.00
16.40	19.60

Results and Discussion

Mercury was found in all 1,000 hair samples, with the average concentration being 4.18 ppm. Mercury levels ranged from a low of 0.12 ppm in a male to a high of 139.0 ppm in a female. These levels in samples were subsequently evaluated according to the sex of the donor and by age groups for each sex (Fig. 1). The average level was higher for females (5.90 ppm) than males (2.45 ppm); similarly, according to the age group, females had mercury levels 1.6- to 3.2-fold higher than those for males. One or more individuals in every age group, however, exceeded the normally expected level of mercury in human hair of 10 ppm (*9*); however, only 20 men (3.37%) had levels above 10 ppm compared with 61 women (14.99%). In two age groups (41 to 60 years and 60+ years), the maximum range approached the 150 ppm mercury level in hair considered dangerous (*9*). The highest level of mercury for males occurred for the age group 11-20 years; males within this age group are statistically the nation's largest food consumers (*11*).

FIGURE 1.—*Mercury residues in hair samples from 1000 Idaho residents by age and sex of donors*



In comparing those individuals with high mercury levels, diet did not seem to play any great role in determining these levels. Generally, individuals with high levels ate little or no fish and, in most instances, no wild game. None of these subjects stated that they were taking drugs containing mercury compounds, and no environmental exposure could be traced to individuals with higher levels. Since few persons with high levels used hair coloring or conditioning preparations, this was also eliminated as a source of mercury exposure. There did not appear to be any specific reason for the higher levels of mercury in females than males, and there was no available data to show evidence of environmental exposure differentially affecting males and females (*5,12-14*).

Sex differences in mercury levels might be influenced by hormonal mechanisms or the biochemical composition of the sexes; however, further studies in this area are needed.

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Comparative Organochlorine Pesticide Residues in Serum and Biopsied Lipoid Tissue: A Survey of 200 Persons in Southern Idaho—1970 [1]

Joe Wyllie, Joe Gabica, and W. W. Benson

ABSTRACT

Paired samples of serum and adipose tissue from patients undergoing abdominal surgery were studied in order to determine residue levels of organochlorine pesticides in the two tissues. Although the residue levels were found to vary with the type of pesticide and with sex and age of the donors, levels in Idaho residents did not differ greatly from persons elsewhere in terms of body burden. Attempts at establishing the degree of compartmental equilibria for p,p'-DDE and p,p'-DDT between the two tissues revealed that the distribution of these two residues was not directly proportional.

Introduction

Beginning with the report of Howell (*1*) in 1948 that DDT residues were present in human fat, the biological storage of persistent insecticides has been studied rather extensively. The literature on human storage was reviewed by Robinson (*10*). The pesticide content of human adipose tissue using samples obtained primarily from autopsy (*2-6*) has been determined for various areas of the world. Fat samples taken from living persons, by biopsy or during routine surgery, have understandably been examined less frequently (*7-9*).

For obvious reasons, blood (and in many cases, serum) has become the tissue of choice for monitoring pesticide residues in humans. However, despite the abundance of available information concerning the respective occurrence of pesticides in blood and fat, relatively little is known of their quantitative relationship to one another. To better understand this relationship, the Idaho Community Study on Pesticides examined 202 paired serum and adipose tissue samples obtained voluntarily from hospitalized patients who lived in the highly agricultural region of southern Idaho.

Sampling Procedures

Both blood and adipose tissue samples were obtained from 141 female and 61 male Caucasian patients undergoing abdominal surgery at Saint Luke's and Saint Alphonsus Hospitals, Boise, Idaho, and Mercy Hospital, Nampa, Idaho. A 10-ml fasting whole blood sample was obtained from each volunteer during the hospital's routine blood collection for biochemistries and approximately 5 g of panniculus fat was taken during the subsequent surgery. In an effort to maintain a maximum degree of validity, only adequately nourished patients were surveyed.

Analytical Procedures

Whole blood was not extracted and analyzed. Because the Community Studies on Pesticides have evaluated serum analysis and use only serum samples in population studies, serum, rather than whole blood, was analyzed for serum-adipose tissue correlations.

Following whole blood centrifugation, serum samples were extracted for organochlorine insecticides by a revised Dale-Cueto triple hexane extraction method (*11,12*).

Two ml of serum was combined with 6 ml of nanograde hexane containing a 20-ng internal standard of aldrin for subsequent recovery determinations and agitated for 3 minutes on a Vortex mixer. The mixture was then centrifuged for 10 minutes at 2,000 rpm and the hexane layer pipetted into a 50-ml concentrator tube. This procedure was repeated three times with unspiked

[1] From the Idaho Community Study on Pesticides, Idaho Department of Health, Statehouse, Boise, Idaho 83707.

hexane, and the combined hexane fractions were then concentrated by means of a modified Snyder column on a steam bath to a final volume of 500 μl.

Adipose tissues were extracted by a modified Mills (*13*) procedure, as follows: the 5-g sample was placed in a mortar and pestle containing 10 g of clean, sharp sand and 1 g of anhydrous sodium sulfate. This mixture was then ground vigorously into a uniform dry granular mass. One ml of nanograde hexane containing 1 μg of methoxychlor as an internal standard was added for subsequent calculation of percent recoveries. The resulting pulverized mixture was transferred to a 150-ml breaker by washing three times with 50-ml petroleum ether. This mixture was then filtered, evaporated to near dryness under nitrogen, cooled to room temperature in a desiccator, and reweighed for percent of fat content. Two grams of this fat weighed on an analytical balance was transferred to a 125-ml separatory funnel where partitioning, extractions with hexane, and subsequent column fractionation were carried out following the procedures previously reported by Mills (*13*) and Mills, Onley, and Gaither (*14*).

The concentrated hexane extracts of sera and fat were injected in 5-μl aliquots into a MicroTek 220 gas chromatograph equipped with two different columns and tritium foil electron capture detectors. The operating analytical parameters were as follows:

Columns:	¼" x 6' glass columns, packed with 1.5% OV-17 and 1.95% QF-1 on 100/120 mesh Chromosorb W, DMCS, HP or 4% SE-30 and 6% QF-1 on 80/100 mesh Chromosorb W, DMCS, HP	
Temperatures:	Column	220° C
	Injection chamber	220° C
	Detector	205° C
Carrier gas:	Nitrogen	
Flow rate:	OV-17/QF-1—70 ml/min	
	SE-30/QF-1—100 ml/min	

The two gas chromatographic columns used have a complete capability of separating for both quantitation and identification of the three isomers alpha-, beta-, and gamma- BHC and many other pesticides not before separable by a one-column determination.

All quantitation of pesticide residues was based on relative peak heights. Each fifth fat sample extract and pooled sera were qualitatively analyzed by thin layer chromatography for confirmation of pesticides reported. Recovery for the aldrin spike in sera and the methoxychlor spike in adipose tissues was 60-90% and 75-95%, respectively; results were corrected to 100%.

Results and Discussion

Seven organochlorine residues (*p,p'*-DDT, *o,p'*-DDT, *p,p'*-DDD, *p,p'*-DDE, dieldrin, β-BHC, and heptachlor epoxide) were present in both fat and serum samples. The means, ranges, and percent occurrence of these residues in samples are given in Table 1. *P,p'*-DDE was the most prevalent and the most highly concentrated compound in both tissues followed by *p,p'*-DDT. Although dieldrin was the third most prevalent residue in sera, it was the sixth most frequently detected in adipose samples. This finding for dieldrin supports the conclusions of Morgan and Roan (*5*) that dieldrin might be more effectively stored in nonlipid tissues. The remaining four compounds occurred quite frequently in adipose tissue, but were detected much less often in serum. The current findings in sera differ slightly from a previous study (*16*) of pesticide levels in sera from 1,000 persons from this region of Idaho in that mean serum *p,p'*-DDE concentrations are somewhat lower (15.5 ppb, this study versus 22.0 ppb) in the previous study and dieldrin in sera was detected much more frequently in this study (88%, this study versus 33%). The present results for pesticide residues in fat generally agree quantitatively with those of recent investigations in other geographic areas, with the possible exceptions of Hawaii (*2*) which reported lower *p,p'*-DDE mean fat levels and Holland (*17*) where studies with adipose tissue revealed *p,p'*-DDE at mean concentrations of only 1.7 ppm. Overall, however, residues in persons in Idaho appear to be comparable to those found elsewhere.

TABLE 1.—*Residue levels of organochlorine pesticides in serum and adipose tissue—Idaho, 1970*

Compound	Serum			Adipose Tissue		
	Residues in ppb		Percent Occurrence	Residues in ppm		Percent Occurrence
	Mean	Range		Mean	Range	
p,p'-DDE	15.5	2-70	100	7.2	0.2-30	100
p,p'-DDT	4.0	0-14	98	1.9	0.1-6.6	100
Dieldrin	0.9	0-10	88	0.2	0-0.7	87
β-BHC	0.3	0-15	23	0.3	0-2.5	98
p,p'-DDD	0.1	0- 4	12	0.1	0-1.2	75
Heptachlor epoxide	0.1	0- 2	18	0.1	0-1.3	97
o,p'-DDT	<0.1	0- 2	4	0.1	0-1.0	89

Concentrations of the four most commonly occurring pesticide residues in serum and adipose tissue (*p,p'*-DDE, *p,p'*-DDT, dieldrin, and β-BHC), as a function of sex of the persons sampled, are shown in Table 2. In both serum and fat, males were found to have significantly higher average levels of *p,p'*-DDE. Males also had higher average levels of *p,p'*-DDT than females in both serum and adipose tissue, but the difference was narrower in adipose tissue. Both sexes had comparable respective levels of dieldrin in both tissues. Mean β-BHC levels in sera were considerably higher in males than females, but mean levels in fat were the same for both sexes. The detection of higher pesticide levels in men agrees with previous findings in Idaho (*16*) and other locations (*6,18*), but is at variance with the results

TABLE 2.—*Residue levels of p,p'-DDE, p,p'-DDT, dieldrin, and β-BHC in serum and adipose tissue by sex of persons sampled—Idaho, 1970*

	Compound	Males (N = 61)			Females (N = 141)		
		Mean	Range	Percent Occurrence	Mean	Range	Percent Occurrence
Serum (Residues in ppb)	p,p'-DDE	20.5	3-70	100	13.4	2-65	100
	p,p'-DDT	4.9	0-14	99	3.7	0-14	98
	Dieldrin	1.0	0-3	92	0.9	0-10	85
	β-BHC	0.6	0-15	26	0.2	0-7	20
Adipose Tissue (Residues in ppm)	p,p'-DDE	8.5	0.2-23	100	6.6	0.3-30	100
	p,p'-DDT	2.1	0.1-7	100	1.8	0.2-7	100
	Dieldrin	0.2	0-1	84	0.2	0-1	89
	β-BHC	0.3	0-3	97	0.3	0-2	99

TABLE 3.—*Residues levels of p,p'-DDE, p,p'-DDT, dieldrin, and β-BHC in serum and adipose tissue by age of persons sampled—Idaho, 1970*

	Compound	Age 0-20 (N = 16)		Age 21-40 (N = 47)		Age 41-60 (N = 80)		Age 61-90 (N = 59)	
		Mean	Percent Occurrence	Mean	Percent Occurrence	Mean	Percent Occurrence	Mean	Percent Occurrence
Serum (Residues in ppb)	p,p'-DDE	7.57	100	14.27	100	16.11	100	17.87	100
	p,p'-DDT	2.55	100	4.46	98	3.92	98	4.52	98
	Dieldrin	.56	94	.88	83	1.06	89	.85	86
	β-BHC	.09	13	.34	21	.55	25	.36	20
Adipose Tissue (Residues in ppm)	p,p'-DDE	3.47	100	6.73	100	7.06	100	8.72	100
	p,p'-DDT	1.01	100	1.91	100	1.67	100	2.08	100
	Dieldrin	.14	100	.13	83	.17	88	.16	86
	β-BHC	.15	94	.24	100	.36	98	.34	97

of Fiserova-Bergerova *et al.* (*19*), who found no differences, as a function of sex, with respect to *p,p'*-DDT-derived materials in a group of 71 people in Florida. This tendency for males to store organochlorines at higher levels may be due in part to the relative complexity of female hormonal interrelationships which could conceivably result in increased microsomal enzyme activity and subsequent body burden reduction. The differing female pattern of fat deposition and the likelihood of males having greater environmental exposure to pesticides may also be contributing factors.

Residue levels of *p,p'*-DDE, *p,p'*-DDT, dieldrin, and β-BHC in serum and adipose tissue by age are shown in Table 3. Persons under 20 years of age had the lowest mean concentrations of all four pesticides in both sera and adipose; however, *p,p'*-DDE was the only residue that showed a consistent progression from the lowest level in people below 20 to its highest mean levels in persons ages 61 to 90 years. The tendency for *p,p'*-DDE levels to increase with age is what might be expected, and analysis of variance confirms this—$p = .01$ in serum and $p < .01$ in tissue. Mean levels of *p,p'*-DDT increased significally in both tissues between age groups 0-20 and 21-40 years; however, there was no significant difference among the age groups 21-40, 41-60, and 61-90 years. This trend also carried a statistical significance in serum of $p = .05$ and in tissue, $p = .05$.

Both dieldrin and β-BHC had similar patterns, i.e, there were no significant differences between the mean values found in the different age groups for either of these compounds. The means of positive findings are low for each compound in the total sampling, but the levels found varied widely. The statistical evaluation of this is negative. In Florida (*19*) dieldrin levels in fat were reported to increase considerably in persons over 20 years of age; this could be due to differences in patterns of pesticide application and analytical techniques.

The increase of *p,p'*-DDE in man with age may be expected in terms of *in vivo* organochlorine catabolism, according to the generally accepted scheme of Peterson and Robinson (*20*). DDT is dechlorinated in the body to DDD which then degrades to the excretable DDA or is excreted as DDT. DDE storage is not appreciably derived from ingestion of DDT, however, but by ingestion of DDE previously broken down in the environment from DDT (*15,21*). Failure of DDE to be effectively eliminated would then result in body burden levels that increase with age. DDT, in contrast, would be broken down and excreted much more readily, and thus, tissue levels would not be expected to increase so dramatically with time.

The considerable difference between serum and adipose tissue in the relative frequencies of occurrence of *p,p'*-DDD, *o,p'*-DDT, β-BHC, and heptachlor epoxide

(Table 1) probably reflects the limitations of present analytical capabilities using serum samples of this size, rather than implying that these residues are sometimes present at high levels in fat but absent from serum. However, since *p,p'*-DDE and *p,p'*-DDT were detectable in both tissues in nearly all cases (Table 1), it would seem reasonable to correlate their respective tissue levels in an attempt to assay the degree of proportionality between serum and adipose tissue residues. This was done by plotting the individual mean serum levels of *p,p'*-DDE and *p,p'*-DDT against their corresponding values for fat. From this information, Pearson correlation coefficients (r) were then derived. These point distribution studies are portrayed graphically in Fig. 1 through 4. It appears from these data that *p,p'*-DDE and *p,p'*-DDT levels in serum and adipose tissue from the persons sampled are not directly proportional, although females do seem to show a more positive correlation (Fig. 1 and 3) between levels in the two tissues than do males (Fig. 2 and 4). However, differences could be due to the occupationally and sexually related factors previously cited, as well as the smaller number of males sampled.

Barquet *et al.* (*18*) have postulated that serum and whole blood titres reflect both the DDE- and the total DDT-derived contents of adipose tissue; this assumption was based on two studies consisting of 59 and 65 persons each. This relationship has also been proposed by Radomski *et al.* (*22*) who examined 20 specimens of whole blood and adipose tissue obtained from autopsy sources and reported pesticide levels between the two tissues to be highly proportional. In contrast, we must tentatively conclude that lipid stored organochlorines are not always accurately predictable on the basis of serum levels. It thus appears that although serum residues may provide a convenient and useful estimate of acute exposure, their validity as an index of chronically acquired adipose body burden may be somewhat limited. To be sure, samples obtained from hospital

FIGURE 1.—*Distribution of p p'-DDE residue levels in serum and adipose tissue from 141 females*

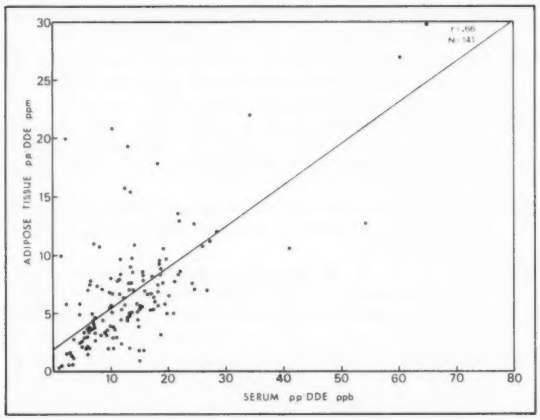


FIGURE 2.—*Distribution of p,p'-DDE residue levels in serum and adipose tissue from 61 males*

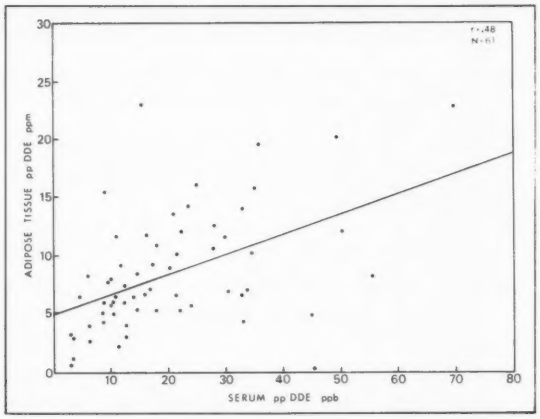


FIGURE 3.—*Distribution of p,p'-DDT residue levels in serum and adipose tissue from 141 females*

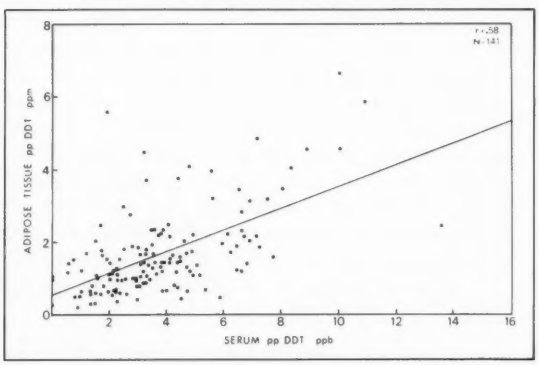
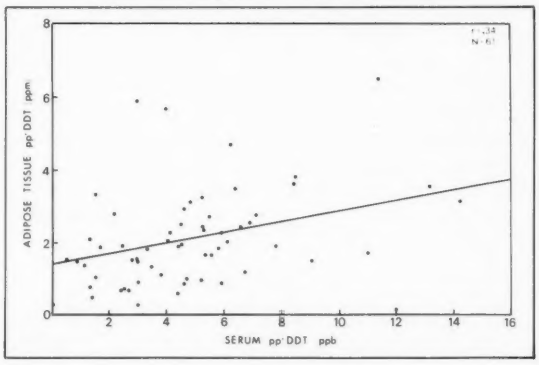


FIGURE 4.—*Distribution of p,p'-DDT residue levels in serum and adipose tissue from 61 males*



patients awaiting surgery are subject to some bias; for example, pathologically and chemotherapeutically induced biochemical changes could conceivably affect pesticide metabolism in such persons. However, the difficulties involved in obtaining adipose biopsies of sufficient size and number from the general population makes the use of hospital specimens a more practical choice at this time.

See Appendix for chemical names of compounds discussed in this paper.

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RESIDUES IN FOOD AND FEED

Arsenic Residues in Soil and Potatoes From Wisconsin Potato Fields—1970 [1]

D. R. Steevens, L. M. Walsh, and D. R. Keeney

ABSTRACT

Potato fields in Wisconsin known to have been treated with sodium arsenite ($NaAsO_2$) were surveyed in 1970 to determine residue levels of arsenic (As) in potato tuber peelings and flesh and in the soil. Total soil As residues ranged from 2.2 to 25.7 ppm and were generally related to the amounts applied. Potato tuber peelings contained 0.2 to 2.6 ppm As, but regardless of the amount of $NaAsO_2$ applied, the tuber flesh did not exceed 0.6 ppm As. It was concluded that As had not accumulated in these Wisconsin potato fields to potentially harmful levels.

Introduction

Accumulation of arsenic (As) in soil as a result of repeated use of pesticides containing As in orchards, cotton, and tobacco fields has been found to adversely affect the quality and quantity of crops grown (*1,4*). Studies to investigate the possible toxic effects and plant uptake of As applied to a Plainfield sand in Wisconsin (*3,6*) showed that phytotoxic effects to vegetable crops did not occur until 90 kg of As per hectare (ha) or more had been applied. Residual phytotoxicity remained for four cropping seasons after application of As at rates of 90 to 720 kg/ha. Except for potato peelings, edible portions of vegetable crops were not contaminated with As. In view of this finding and the fact that sodium arsenite had been used extensively in Wisconsin as a potato vine defoliant (applied at about 9 kg As/ha) from 1950 to 1968, this study was undertaken to survey commercial potato fields in Wisconsin to ascertain if excessive levels of As had accumulated in the soil and potato tubers (flesh and peelings) due to use of sodium arsenite.

Materials and Methods

In August 1970, soil and potato tuber samples were collected from a total of 18 potato fields in the three principal potato-growing areas of Wisconsin. At each site, a composite soil sample consisting of 10 randomly selected cores was taken from the plow layer. A composite potato tuber sample was also obtained at each site by taking tubers from five potato hills selected at random. The soil samples were air-dried (60°C) and ground (< 80 mesh). Potato tuber samples were scrubbed using distilled water and a nylon fiber brush. Potato tuber peelings and flesh were analyzed separately; samples were air-dried (60°C) and ground (20 Mesh).

For total As analyses, plant tissue samples were digested by using the HNO_3-$HClO_4$ procedure of Blanchar *et al.* (*2*), and soil samples were digested by the H_2SO_4-$HClO_4$ procedure of Small and McCants (*5*) with modification made by Jacobs *et al.* (*3*). This procedure gave quantitative (>95%) recovery of 10-100 ppm As added as As_2O_3 to soils. In addition, the amount of available soil As was determined by extracting with Bray P-1 solution (0.025N HCl and 0.03N NH_4F), an extractant commonly used to determine available soil phosphorus (*3*). Arsenic in the potato and soil digests and soil extracts was determined by the reduction-distillation method of Small and McCants (*5*) with modification made by Jacobs *et al.* (*3*). The sensitivity limit for this method was 0.1 ppm of As.

Results and Discussion

Results of analyses of soil and potato tubers are reported in Table 1; arsenic residues reported represent the average of analyses of duplicate samples.

[1] From the Department of Soil Science, Univ. of Wisconsin, Madison, Wis. 53706.

In general, total soil As levels were related positively with As application as reported by growers, although wide variations were noted (Table 1). Part of this disparity is probably due to the fact that the stated amount of As application was usually only an estimate. The level of naturally occurring As in soils is considered to be from 2 to 10 ppm (*1,4*). In this survey, As levels in the soil were <10 ppm at 7 sites and 10.0 to 25.7 ppm at 11 sites. The available Bray P-1 extractable As in soil was less than 5 ppm for all sites sampled. In comparison with the data obtained in experimental field plots (*3,6*), soil As levels were below any concentration deterimental to crops.

The As concentration in potato peelings taken from the survey sites ranged from 0.2 to 2.3 ppm, while the greatest As level in the flesh was 0.6 ppm. The As tolerance limit for several vegetable crops has been established at 2.6 ppm (*7*); all potato samples obtained in this survey were well below this limit, especially considering that the peelings constitute only a small portion of the total potato tuber.

Sodium arsenite was removed from registration as a potato vine defoliant in 1969. The results of this survey indicate that past usage of $NaAsO_2$ has not resulted in levels of As in Wisconsin soils which would be phytotoxic or which would cause harmful levels of As to accumulate in potato tubers.

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TABLE 1.—*Amount of arsenic applied as $NaAsO_2$, total and extractable Bray P-1 arsenic in soil samples, and arsenic concentration in potato tuber peelings and flesh from Wisconsin potato fields—1970*

Type of Soil	Sampling Sites	Soil			Potato	
		Total Arsenic Applied (kg/ha)[1]	Total Arsenic (ppm)[2]	Bray P-1 Extractable Arsenic (ppm)[2]	Peelings (ppm)[2]	Flesh (ppm)[2]
Sandy loams						
(Northwestern Wisconsin, Spooner area)	1	56	8.0	2.4	1.1	.4
	2	32	6.0	1.0	.5	.2
	3	32	9.8	2.8	1.6	.2
	4	25	17.4	2.9	1.8	.2
Silt loams						
(North-central Wisconsin, Antigo area)	5	65	25.7	1.4	.2	T
	6	20	5.4	.6	.2	T
	7	96	25.9	4.6	.5	.1
	8	25	10.4	1.0	.4	.2
	9	8	13.9	1.6	.4	.4
	10	48	16.0	1.1	.4	.2
	11	25	10.0	.5	.5	.4
Loamy sands						
(Central Wisconsin, Stevens Point area)	12	40	8.6	1.0	1.6	.4
	13	53	14.9	1.9	1.6	.4
	14	28	10.1	1.2	1.3	.6
	15	48	12.3	1.6	2.3	.1
	16	40	10.4	1.7	1.8	T
	17	0	2.2	.3	.4	T
	18	16	5.7	1.0	1.3	T
Plainfield sand						
[Data from a previous study in Wisconsin (*6*)]	—	0	3.6	.4	2.4	.1
	—	45	12.5	2.0	5.2	.1
	—	90 [3]	24.3	5.2	9.8	.1
	—	180 [3]	45.0	10.1	22.2	.5
	—	720 [3]	121.0	25.2	53.7	.6

NOTE: T = trace = <.1ppm.
[1] Growers' estimate of total arsenic applied to sites 1-18.
[2] Average of duplicate samples.
[3] Phytotoxicity was noted in studies (*3,6*) after application of arsenic at rates of 90-720 kg/ha.

RESIDUES IN FISH, WILDLIFE, AND ESTUARIES

Mercury Concentrations in Game Birds, State of Washington—1970 and 1971 [1]

Frank E. Adley and Donald W. Brown

ABSTRACT

In a survey to determine the presence of mercury in the liver of game birds, 246 specimens contributed by local hunters during October 1970 to January 1971 and 4 obtained in May 1971 were analyzed. The species included pheasant, quail, chukar partridge, duck, geese, and grouse. Muscle samples were submitted with some of the liver specimens and, in those instances, these tissues also were analyzed.

The highest average mercury concentrations were in livers from mergansers (11.67 ppm) and teal (0.29 ppm), species with diets including aquatic organisms and thus differing from the other samples. The average mercury levels in liver from the other species were 0.08 ppm in pheasant; 0.03 ppm, quail; 0.06 ppm, chukar partridge; 0.12 ppm, ducks (other than teal and mergansers); 0.16 ppm, geese; and 0.02 ppm, grouse.

Of the species with both muscle and liver tissue analyzed, quail had the highest muscle to liver ratios of mercury and mergansers, the lowest.

In view of the practice of fall seeding with mercury-dressed seeds, the importance of the period of exposure to the treated seeds was investigated. No apparent correlation was evident between liver concentrations and the time of year the birds were killed.

Introduction

Relatively recent findings of elevated mercury levels in ducks from the Lake St. Clair-Detroit River area of Michigan (*1*) prompted investigations by several States and Canadian Provinces to determine levels of mercury in their wildlife. In some instances, (*1*) the concentrations of mercury were found to exceed the guideline limit of 0.5 ppm of mercury set by the Food and Drug Administration for edible portions of fish, and, in 1969, the pheasant season in the Province of Alberta was closed due to excessive mercury levels. In the interest of game management, the State of Washington Department of Game instigated a study to monitor pheasant in certain areas where mercury-treated seeds were being used (Burton J. Lauckhart, *personal communication*).

Mercury concentrations above 0.5 ppm were detected in several pheasant livers and one breast; however, from an evaluation of the overall findings, it was concluded that mercury residues were not present at levels that would constitute a hazard to consumers.

To supplement this study, the Hanford Environmental Health Foundation of Richland, Wash., initiated a program to monitor, during the 1970-71 hunting season, other field-killed game birds native to the Columbia Basin, which is a consistently popular hunting area. Specimens were contributed largely by sportsmen and field personnel of the State Department of Game and were from the area of the Basin from Ephrata to the north down to Paterson, and from Prosser on the west to Dayton, a productive upland and migratory bird hunting area of about 8,000 square miles.

In view of the practice of fall seeding with mercury-dressed seeds in this region, the study was designed to detect any apparent increase in body burden of mercury as the season progressed. Thus, samples were collected from the beginning of the hunting season in October through the close of the hunting season in January.

Sampling Procedures

Uniform instructions for sampling were disseminated to those who would have specimens to contribute. It was requested that samples of whole liver be placed in plastic bags and kept frozen for delivery to the laboratory. Essentially all samples were identified with the following information: location and date of kill, type and sex of bird, statement as to general age, and name of submitter.

[1] From the Hanford Environmental Health Foundation, Inc., P. O. Box 100, Richland, Wash. 99352.

Analytical Procedures

Samples accurately weighed to .5 g were dissolved in 1:1 sulfuric-nitric acids at about 60° C. On dissolution, the samples were cooled and a 6% w/v solution of potassium permanganate was added until purple color persisted. This was made to 100-ml volume, and an aliquot was analyzed by flameless atomic absorption spectrophotometry. In this procedure (*2,3*) the mercury compounds are reduced to elemental mercury by the addition of stannous chloride, and the mercury is swept by an air stream through an absorption cell mounted on the burner of an AA unit. The resulting absorption is measured and compared to a calibration curve. This procedure has a sensitivity of 0.01 ng of mercury as mercuric chloride per sample per 1% absorption. Reproducibility was determined to be better than ± 10% using mercuric chloride standard.

Results and Discussion

During the season, over 300 specimens were received by the laboratory. Of these, 250 were selected, Table 1, which could provide the best coverage of the parameters selected for investigation. These included four mergansers collected in May 1971 as part of a Federal monitoring program. All others were collected between October 1970 and January 1971.

The results of mercury analysis of liver samples by month of collection are given in Table 2. For the 85 pheasant analyzed, the mercury concentration averaged 0.08 ppm; three of these samples had concentrations exceeding 0.5 ppm, the highest being 0.71 ppm. The average level in the 64 ducks (mostly mallard and excluding teal and mergansers) was 0.12 ppm with none exceeding 0.39 ppm. The mercury level for the 15 geese averaged 0.16 ppm; for the 13 chukar partridge, 0.04 ppm; the 10 quail, 0.03 ppm; and the 1 grouse, 0.02 ppm. The highest average levels were in the 6 mergansers and 10 teal, 11.67 ppm and 0.29 ppm, respectively.

The higher values for mergansers and teal are interesting since they differ in diet from the other birds. The normal diet for mergansers consists of small fish, insects, and crustacea, and the diet of the teal may include certain aquatic life having an elevated methylmercury content.

There appeared to be no marked increase in body burden of mercury through the hunting season, October through January, although ducks (excluding teal and mergansers), chukar, and quail did show higher levels in the latter part of the season (Table 2).

In some instances, muscle tissue was also submitted with liver samples. The analytical findings are shown in Table 3. Quail and geese had the highest muscle to liver ratios, whereas the mergansers had consistently

TABLE 1.—*Upland and migratory birds analyzed—Washington, 1970-71*

SPECIES	NUMBER ANALYZED
Pheasant	85
Ducks	
Teal [1]	10
Mergansers [1]	6
Other	64
Geese	15
Chukar Partridge	59
Quail	10
Grouse	1
TOTAL	250

[1] Separated due to eating habits.

TABLE 2.—*Mercury concentrations in livers of birds by month of collection*

SPECIES	DATE OF COLLECTION	NUMBER ANALYZED	AVERAGE Hg CONCENTRATION (PPM)	RANGE (PPM)
Pheasant	October	35	0.05	0.01-0.15
	November	23	0.12	0.01-0.71
	December	27	0.09	0.003-0.45
	Total	85	0.08	0.003-0.71
Ducks				
Teal [1]	October	6	0.39	0.009-0.47
	November	4	0.13	0.01-0.17
	Total	10	0.29	0.009-0.47
Mergansers [1]	November	2	31.10	4.15-58.00
	May 1971 [2]	4	1.97	0.80- 3.40
	Total	6	11.67	0.80-58.00
Other	October	16	0.07	0.03-0.08
	November	10	0.07	0.02-0.11
	December	14	0.12	0.06-0.17
	January	24	0.16	0.07-0.39
	Total	64	0.12	0.02-0.39
Geese	November	7	0.226	0.14-0.34
	December	7	0.097	0.05-0.16
	January	1	0.08	—
	Total	15	0.16	0.05-0.34
Chukar Partridge	October	39	0.03	0.02-0.09
	November	7	0.05	0.02-0.21
	December	13	0.06	0.03-0.11
	Total	59	0.04	0.02-0.21
Quail	October	5	0.02	0.01-0.02
	November	5	0.03	0.01-0.07
	Total	10	0.03	0.01-0.07
Grouse	October	1	0.02	—

[1] Separated due to eating habits.
[2] Federal monitoring sample.

TABLE 3.—*Muscle/liver mercury concentration ratios by species*

SPECIES	MUSCLE/LIVER RATIOS
Pheasant	0.65, 0.75
Quail	3.0, 2.5, 1.0, 1.0
Geese	0.8, 1.25, 3.8, 0.7
Mergansers	0.20, 0.16, 0.20, 0.21

low muscle to liver ratios. It is possible that mergansers, due to their diet of aquatic life high in the food chain, possess relatively less mercury in their edible muscle tissue than in liver tissue.

The pheasant is one of the most important upland bird species since over 200,000 pheasants are harvested annually by hunters in the Columbia Basin area (*4*). During the 1969-70 winter season, the State of Washington Department of Game monitored 37 wild pheasants taken as road kills from Eastern Washington (Burton J. Lauckhart, *personal communication*). The average mercury content of liver in this group was 0.24 ppm. Since that time ranchers, seed producers, and the Federal and State governments have taken steps to minimize the use of mercury-bearing seed dressings. It is noteworthy that the findings of this study reflect a reduction in the mercury content of pheasants from an average of 0.24 ppm in 1969-70 to 0.08 ppm in 1970-71, which may be attributed to a general reduction under way in the use of mercury-dressed seeds.

Acknowledgment

This study could not have been performed without the cooperation of the members of the Richland Rod and Gun Club and other interested sportsmen. The advice and cooperation of the Division of Game Management, Washington State Department of Game, and publicity by radio station KONA, were also helpful. The authors gratefully acknowledge all those who made this work possible.

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(*3*) *Rathje, A. O. 1969.* A rapid ultraviolet absorption method for the determination of mercury in urine. Am. Ind. Hyg. Assoc. J. 30(2):126-132.
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Effects of Estuarine Dredging of Toxaphene-Contaminated Sediments in Terry Creek, Brunswick, Ga.—1971 [1]

Charles J. Durant and Robert J. Reimold

ABSTRACT

The possible reintroduction of toxaphene into estuarine biota from dredging and displacement of contaminated sediment in Terry Creek, Brunswick, Ga., was studied. In the estuary, the sediments near a toxaphene plant outfall were found to be contaminated with toxaphene approaching 2,000 ppm, and oysters collected 2 miles from the outfall were found to contain residue levels near 6 ppm. Analyses of oysters and sediment before and after dredging operations revealed no significant increase of toxaphene residues resulting from the dredging and resultant spoil runoff.

Introduction

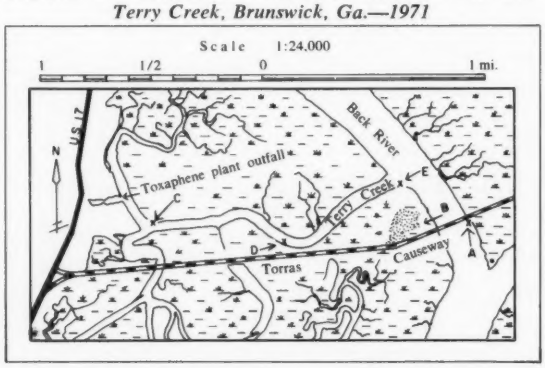
Terry Creek, Brunswick, Ga., is an estuarine watershed through which industrial wastes have been discharged to the ocean for the past 50 years; among the contaminants introduced over the past 20 years have been the manufacturing residues and wastes of toxaphene. Previous surface and subsurface sediment core samples had revealed high concentrations of toxaphene in the sediments. It was reasonable to assume that a proposed dredging operation from June 10-13, 1971, to widen and deepen the existing channel of Terry Creek (Fig. 1) would release concentrations into the water and that the toxaphene would be incorporated into the biota, perhaps causing mortality. The purpose of this study was to determine levels of toxaphene in nearby oysters and sediment, measure any increase in these levels, and assess any ecological damage caused by the dredging operation.

Sampling Procedures

The oyster (*Crassostrea virginica*) was used as an indicator of contamination because it is a sessile filter feeder with the ability to concentrate pesticides up to 70,000 times that found in the surrounding water (*1*). Oysters were collected at the Torras Causeway toll bridge crossing Back River (Fig. 1, location A) weekly from April 1 to June 23, 1971, and monthly from then until October 12, 1971. Each sample was taken from as near as possible to mean low water and included a minimum of 12 shucked oysters which were placed in a mason jar.

Three sediment core samples were taken from Terry Creek on June 10, 1971, one each from locations C, D, and E (Fig. 1). Location C was a point on the north shore of Terry Creek, 50 yd from the junction with Dupree Creek and .2 mile from the toxaphene plant outfall; location D was on the south shore of Terry Creek, .8 mile from the plant outfall and one-half the distance

FIGURE 1.—*Geographic locations of collection sites near Terry Creek, Brunswick, Ga.—1971*



[1] Contribution 330 from the Marine Institute, The University of Georgia, Sapelo Island, Ga. 31327.

from the outfall to Back River; and location E was on the south shore of Terry Creek, 1.4 miles from the plant outfall and 50 yd west of Back River. Each sample (70 mm diameter, 80 cm long) was collected with a clear plastic core barrel. Samples were extruded from the liner in 10-cm increments by forcing the sediment out with a tightly fitting rubber plunger. In addition to the core samples, a composite sediment sample consisting of six surface grab samples was collected from a transect across the dredge spoil area (location B, Fig. 1) on June 23, 1971, about 10 days after dredging ceased.

In addition to sampling, field observations of biota were made in the vicinity of the dredging operation on June 10 and 11.

Analytical Procedures

Analytical procedures generally followed those reported by Wilson (*2*). Each oyster sample (12 specimens) was homogenized in an Osterizer, and a 30-g aliquot of the homogenate was then mixed with a desiccant consisting of 9 parts anhydrous powdered sodium sulfate and 1 part Quso (a micro fine precipitated silica). The mixture was alternately frozen and blended until a free-flowing powder was obtained. The samples were then Soxhlet extracted for 4 hours with glass-distilled petroleum ether, concentrated and partitioned with acetonitrile, and evaporated to dryness at room temperature. The residue was eluted from a Florisil column with 6% ethyl ether in petroleum ether.

Sediment samples were processed as follows: Each 10-cm increment in the core was placed in a separate beaker and thoroughly mixed. An aliquot of each sample was thinly spread in the bottom half of an open petri dish and dried in darkness at ambient temperature for a minimum of 5 days. A 5- to 15-g aliquot of the dried sediment was mixed with a desiccant in a 1:3 ratio by weight. The extraction and cleanup procedures were identical to those previously described for oyster samples.

The oyster and sediment extracts were then quantified on Varian model 600 D gas chromatographs equipped with tritium electron capture detectors and glass columns (5′ by ⅛″, o.d.) packed with 3% DC-200. Operating parameters were: oven temperature—193° C; injector and detector temperatures—210° C; and carrier gas—N_2 at 20 ml/min.

For confirmation, extracts were injected on a mixed column of 5% QF-1 and 3% DC-200 (1:1 by weight) with the same operating parameters as above.

Recovery rates of toxaphene in oysters and in sediments were above 85% and 90%, respectively. Toxaphene concentrations below 0.25 ppm were considered insignificant for the purpose of this study. The data were not corrected for percent recovery.

Results and Discussion

The results of toxaphene analyses of oyster samples are presented in Fig. 2. The mean of the weekly concentrations in May was 3.30 ppm. Oysters collected on June 10, 1971, at about 10 a.m. on the day the dredging operation began, had a higher concentration than subsequent samples but contained 1.24 ppm less than the May average.

The results of toxaphene analysis of the sediment core samples are summarized in Table 1. The concentrations in the first 10-cm increment (surface to 10-cm depth) of the three samples ranged from 35.5 to 1,858.3 ppm. The composite sample of surface sediment obtained after dredging (site B) contained 32.8 ppm toxaphene on a dry-weight basis.

Analyses of oysters and sediment (before and after dredging operations) revealed no significant increase of toxaphene residues resulting from the dredging and resultant spoil runoff.

During the course of field observations, a few anchovies (which are also filter feeders) were observed in distress as a result of the dredging operation. The cause appeared to be related to the heavy suspended sediment

FIGURE 2.—*Concentration of toxaphene (in ppm, wet-weight basis) in the American oyster* (Crassostrea virginica) *collected from near the intersection of Terry Creek with Back River, Brunswick, Ga. (as depicted in Fig. 1)*

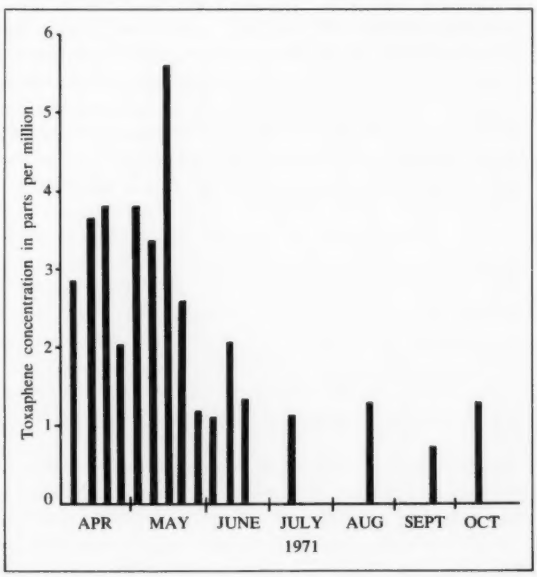


TABLE 1.—*Toxaphene concentrations in three sediment cores, by 10-cm increments, collected from Terry Creek, Brunswick, Ga.—June 10, 1971*

Sampling Location	Residues in PPM and Content of Sediment							
	Surf. to 10 cm	10-20 cm	20-30 cm	30-40 cm	40-50 cm	50-60 cm	60-70 cm	70-80 cm
Site C [1]	1,858.3 mud & some chips	1,340.5 mud & few chips	1,324.0 mud	1,367.2 mud	1,236.7 mud	433.6 mud	68.5 mud	83.2 mud
Site D [2]	111.85 mud & few chips	615.64 mud & many chips	16.04 mud	17.46 mud	5.42 mud	3.4 mud	2.88 mud	—
Site E [3]	35.5 mud	35.47 mud	21.9 mud	70.65 mud	79.8 mud	21.0 mud	18.5 mud	5.27 mud

NOTE: — = no sample taken.
[1] North shore of Terry Creek, 50 yd from junction with Dupree Creek and .2 mile from toxaphene plant outfall.
[2] South shore of Terry Creek, .8 mile from the plant outfall and one half the distance from the outfall to Back River.
[3] South shore of Terry Creek, 1.4 miles from the plant outfall and 50 yd west of Back River.

load in the water which clogged the gill cavities of the fish, probably causing them to suffocate, rather than related to pesticide poisoning. Several weeks prior to the Terry Creek operation, menhaden and anchovies were observed in distress with clogged gill covers in the vicinity of another dredging operation in the Intracoastal Waterway just north of Altamaha Sound. No lethal concentrations of pesticides were suspected in the Altamaha Sound area, since oysters collected there at monthly intervals for several years have contained no more than trace amounts of pesticides and no toxaphene.

The anticipated sudden kill of fish and shellfish from toxaphene poisoning in Terry Creek did not occur, probably for a combination of reasons: (1) There was less runoff of toxaphene contaminated sediments than expected from the dredging operation. (2) The toxaphene was bound to the clay particles and therefore not available to the biota. (3) The toxaphene in the lower sediments (below 10 cm) had undergone degradation and detoxification; chromatograms of sediments showed a marked difference with depth (illustrated by a buildup of peaks in the first portion of the chromatograms and diminishing peaks in the later portion) suggesting *in situ* dechlorination. (4) Lastly, the possibility exists that some fin fish were able to avoid the area in question.

Any future dredging activities should be planned and coordinated with further research. The relation of suspended sediment load to toxaphene concentration should be of primary concern. Additional research is needed to document whether or not sediment toxaphene concentration increases in the direction from the mouth of Terry Creek toward the toxaphene plant outfall (Fig. 1). Detoxification studies in the form of static bioassays using toxaphene extracted from core sediments are currently in progress.

See Appendix for chemical name of toxaphene.

This work was funded in part by a grant from Hercules, Inc. to the University of Georgia.

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Organochlorine Insecticide Residues in Water, Sediment, and Organisms, Aransas Bay, Texas—September 1969 - June 1970

Roger R. Fay[1] and Leo W. Newland[2]

ABSTRACT

An investigation was conducted to determine the presence and distribution of organochlorine insecticide residues in Aransas Bay and its contributing bays at Rockport, Tex. A total of 80 water samples, 29 sediment samples, and 11 samples of 8 different types of organisms were collected and analyzed from September 1969 through June 1970.

Organochlorine insecticide residues were detected in only 3 water samples and 3 sediment samples and, although residues were detected in 8 of the 11 organism samples, these were all at low levels (<67 ppb). The predominant residues found in the organisms were dieldrin, p,p'-DDD, and p,p'-DDE. The occurrence or concentration of residues could not be related to salinity or pH of the water or percent organic content of the sediments.

Introduction

The extensive use of organochlorine insecticides has led to an accumulation of these residues in the environment. In estuaries, the persistence of organochlorine residues creates a hazard to fish and other forms of marine life (*3*). Estuaries, in addition to supporting large shellfish populations, provide spawning and nursery grounds for many species of fish. Shellfish larvae and young fish are extremely susceptible to chemical pollutants and other forms of environmental stress. The problem is compounded by the fact that some fish and shellfish are known to concentrate organochlorine residues in their tissues and, thus, pass these residues along the food chain.

Insecticides entering the estuary in water can be introduced into the food chain (1) when residues are absorbed by plankton or other organisms and (2) when residues are adsorbed to suspended particles which settle and become part of the bottom sediment where benthic fauna feed (*5*). Insecticides entering on silt and detritus also may become part of the sediment.

The food chain at the same time plays an important role in removing insecticides from estuaries since fish and porpoise immobilize insecticide residues in their body tissues and, thus, these insecticides are removed from estuaries when fish migrate to sea. Similarly, fish-eating birds accumulate large amounts of insecticides and then distribute them along migratory pathways. The major portion of insecticides entering estuaries, however, are lost through dilution, chemical decay, and adsorption by bottom deposits (*5*).

The effects on aquatic organisms of chronic exposure to organochlorine insecticides were first realized with the reproductive failure of lake trout in New York and high mortalities in hatchery-reared coho salmon related to *p,p'*-DDT residues in the eggs (*2*). Investigations have shown that, of the commonly used organochlorine insecticides, endrin has the highest acute toxicity to fish. Fish subjected to chronic exposure of low concentrations of endrin were found to be more susceptible than control fish to lethal concentrations of endrin during the first 24 hours of exposure; however, by 72 hours the two groups exhibited similar survival (*10*).

Extensive experimentation has been conducted to determine the effects on oysters of chronic exposure to sublethal concentrations of *p,p'*-DDT and other organochlorine insecticides. Under controlled laboratory conditions, oysters exposed to *p,p'*-DDT at concentra-

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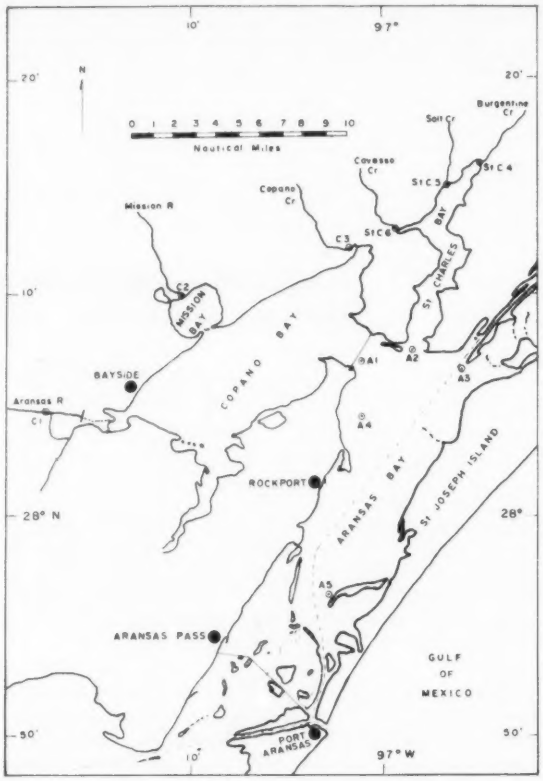
tions of 1 to 2 parts per billion (ppb) at 30° C appeared to grow and behave in a manner similar to that of the control groups; analyses of the tissues, however, revealed concentrations of *p,p'*-DDT many times that of the environment (*4*). A concentration of 7 to 10 ppb *p,p'*-DDT in water inhibited 50% of the normal shell deposition in oysters (*3*). Physiological irritation was shown by spasmodic shell movements when the concentration of *p,p'*-DDT in the water was increased to 0.1 parts per million (ppm), and at a concentration of 1 ppm, the oysters usually remained closed (*4*). In the presence of continuous low level *p,p'*-DDT pollution, oysters concentrated the insecticide in their tissues to more than 25 ppm within 1 week without showing any harmful effects; however, 50% of the small fish and shrimp fed this oyster meat died within 2 days (*4*).

Dissection of oysters containing *p,p'*-DDT residues has shown that 67% of the *p,p'*-DDT was stored in the intestinal tract, digestive gland, and gonads. The gonads were found to be a major site of *p,p'*-DDT storage with residues in excess of 25 ppm reported in the gametes. An attempt was made to culture oyster larvae from such highly contaminated gametes in order to determine the effects of inherent *p,p'*-DDT on oyster development. Unfortunately, the experiment failed at an early stage due to technical difficulties in the laboratory (*4*).

The Gulf Breeze Laboratory of the Environmental Protection Agency (formerly under the Bureau of Commercial Fisheries, USDI) at Gulf Breeze, Fla., is engaged in a monitoring program for organochlorine insecticides. Oyster samples from coastal States are collected at 30-day intervals and analyzed for organochlorine insecticide residues. In 1967, oyster samples were obtained on a monthly basis from 12 stations in Texas. Of the 129 samples analyzed, 90% contained one or more organochlorine insecticide residues (*6*). A similar investigation of Galveston Bay following extensive mosquito control programs in the fall of 1964 (*7*) reported no indication of elevated insecticide levels in water and oyster samples. Insecticide levels were low in both water and oysters, usually less than 0.01 ppm if detected.

The study reported here was conducted to determine the organochlorine residues in Aransas Bay, Tex., and its adjacent contributing bays, Copano, Mission, and St. Charles, (Fig. 1) from September 1969 through June 1970. The land area surrounding Aransas, Copano, and St. Charles Bays is primarily ranch land with secondary agricultural usage. The area is drained by the Aransas and Mission Rivers and the Copano, Cavasso, Salt, and Burgentine Creeks. Samples collected for analysis included water, sediment, and various types of organisms. Locations of sampling stations are shown in Fig. 1.

FIGURE 1.—*The area of investigation showing sampling stations, Aransas Bay, Texas—1969-70*



Collection of Samples

Water samples were collected monthly from September 1969 through June 1970. Samples were taken at the surface in clean 2-qt glass jars with aluminum foil lined lids and stored in darkness to prevent photochemical alteration of insecticide residues.

Sediment samples were collected in September and November 1969, and January, May, and June 1970 using a grab sampler; samples were then refrigerated to minimize loss of the insecticide due to biochemical degradation.

Dates of collection of organism samples listed below depended on their availability; shrimp, oysters, and crabs were obtained in January and March 1970, and the fish samples were collected in July 1969.

sea trout	*Cynoscion arenarius*
drum	*Pogonias cromis*
oysters	*Crassostrea virginicia*
blue crab	*Callinectes sapidus*
silverside minnows	*Menidia menidia*
yellowtail croaker	*Bairdella chrysura*
ribbon fish	*Trichiurus lepturns*
brown shrimp	*Penaeus aztecus*

Each sample represented an average of at least 12 individuals of a species. All organisms were collected with a trawl, except for oysters which were picked from the reef, and then frozen until analysis.

Analytical Procedures

The procedure for quantitatively extracting organochlorine insecticides from water, sediments, and organisms is described below. The solvents, hexane and hexane: acetone (41:59) used in the extraction techniques, were repurified by glass distillation employing a 3-ball Snyder column.

WATER

One liter of water was separated from suspended material by centrifugation or settling; then 500 ml of the sample was extracted with 100 ml of hexane by shaking for 1 minute in a separatory funnel. After separation of the layers was complete, the water was drawn off; the remaining 500 ml of water added, and the sample extracted by shaking for 1 minute. If emulsification was encountered between the hexane and water, anhydrous Na_2SO_4 was added to break the emulsion. The hexane phase was quantitatively transferred to a Kuderna-Danish concentrating apparatus equipped with a 3-ball Snyder column and concentrated in a water bath of 5 ml. The extract was transferred to a 10-ml volumetric flask and brought to volume. The hexane extract was analyzed directly by gas-liquid chromatography (GLC without further cleanup.

SEDIMENT

On removal from refrigeration, the sediment samples were spread and allowed to dry at room temperature. The dry sediments were ground with a mortar and pestle to pass through a #30 mesh sieve, thus insuring uniform saturation of the sample with solvent. A 20- to 30-g subsample (dry weight) was weighed, placed in a Soxhlet extraction apparatus, and refluxed for 6 hours using hexane:acetone as the extracting solvent. Since acetone is unsuitable as a solvent in GLC employing an electron capture (EC) detector, the insecticides were partitioned into the hexane phase by the addition of water. The hexane phase was then transferred to a Kuderna-Danish evaporating apparatus with a Snyder column and concentrated to approximately 10 ml for cleanup by column chromatography.

ORGANISMS

Immediately after thawing, the individual organisms making up a sample were ground to a fine mixture in a blender. A portion of this homogenate was weighed, and three times the subsample's weight of anhydrous Na_2SO_4 was added. The mixture was blended thoroughly to insure adequate distribution of the Na_2SO_4 and adsorption of the water. The sample was extracted following the same procedure as described for sediments.

Following the extraction and partitioning of the insecticides into the hexane phase, the sample was concentrated in the Kuderna-Danish apparatus to approximately 10 ml for cleanup by column chromatography.

CLEANUP OF SAMPLE EXTRACTS

The presence of relatively large amounts of interfering materials in sediments and biological samples requires that these samples be cleaned up prior to analysis by GLC. Column chromatography was selected as a method of cleanup because of its large carrying capacity and its ability to pre-separate insecticide mixtures, thereby aiding in the analysis of subsequent gas chromatograms. Florisil (activated magnesium silicate) was employed as the adsorbent, and elutions were made with 5% ether in hexane and 15% ether in hexane. The procedure followed was essentially that described in reference (9) with modifications (8).

GAS-LIQUID CHROMATOGRAPHY

A Varian Aerograph model 204B equipped with a ^{3}H-foil EC detector was employed for the separation and identification of organochlorine insecticide residues. The use of all glass columns and on-column injection was required to avoid sample degradation resulting from contact of the sample with metal surfaces. Gas chromatographic conditions were as follows:

Columns:	2 m long x 4 mm i.d. glass packed with 10% DC-200 or 10% QF-1 on 60/80 mesh Gas Chrom Q
Temperatures:	Columns: 195° C (QF-1) 215° C (DC-200) Injector: 230° C
Carrier gas:	N_2 with a flow rate of 75 ml/min
Sensitivity:	1.2×10^{-11} amps full scale

Identification was based on retention times from the polar (QF-1) and non-polar (DC-200) columns. The use of two columns of differing polarity decreases the possibility of erroneous identifications. An L&N Speedomax W recorder with a disc integrator was used to calculate peak areas.

All of the samples were analyzed for the following organochlorine insecticides: γ-BHC, heptachlor, heptachlor epoxide, aldrin, dieldrin, endrin, *p,p'*-DDE, *p,p'*-DDD, and *p,p'*-DDT. Analysis was not performed for polychlorinated biphenyls, since the investigation was planned and executed prior to the availability of methods and standards for these compounds.

RECOVERY

Organochlorine insecticide standards of greater than 99% purity were obtained from the following sources: heptachlor, heptachlor epoxide, and endrin from Velsicol Corporation; aldrin and dieldrin from Shell Chemical Company; *p,p'*-DDD from Rohm and Haas; γ-BHC from the U. S. Food and Drug Administration; *p,p'*-DDT from City Chemical Corporation; and *p,p'*-DDE

from the Pesticide Repository, Perrine Primate Laboratory, Environmental Protection Agency. Recovery rates were assumed to be in the range of 90-100% according to the methods used (*1,8,9*).

Results and Discussion

The results of the analysis for organochlorine residues are considered below, followed by an interpretation of these data. An attempt was made to relate the presence or concentration of insecticide residues to the physical and chemical characteristics of the area. Salinity and pH data of the water samples were obtained from the Texas Department of Parks and Wildlife at Rockport, and organic content of the sediments was determined by combustion of the sample at 500° C. A summary of these characteristics of the water and sediment from each sampling station is presented in Table 1.

TABLE 1.—*Salinity and pH of water and organic matter content of sediments at the sampling stations, Aransas Bay, Texas—1969-70*

Sampling Station	Water		Sediments
	Salinity (ppt)	pH	Percent Organic Content (dry-weight basis)
A-1	15.0	8.3	2.1
A-2	12.2	8.3	1.7
A-3	11.1	8.3	14.3
A-4	16.1	8.3	5.4
A-5	20.5	7.9	5.0
C-1	6.6	8.1	5.7
C-2	1.1	8.2	4.7
C-3	8.9	8.0	1.8
St. C-4	11.1	8.4	6.9
St. C-5	12.2	8.3	8.2
St. C-6	8.3	8.4	10.2

TABLE 2.—*Organochlorine residues in water, Aransas Bay, Texas—1969-70*

[T = Trace = <0.1 ppb; — = not detected]

Collection Date	Sampling Location	Residues Detected in ppb (ng/g)
1969		
Sept.	A-1	*p,p'*-DDT (T) endrin (4.4)
	A-2	—
	A-3	—
	A-4	—
	A-5	—
Oct.	A-1	*p,p'*-DDT (T)
	A-2	—
	A-3	—
	A-4	—
	A-5	—
Nov.	A-1	—
	A-2	—
	A-3	—
	A-4	—
	A-5	—
	C-2	—
	St. C-4	—
	St. C-5	—
	St. C-6	—
Dec.	A-1	—
	A-2	—
	A-3	—
	A-4	—
	A-5	—
	C-1	heptachlor (T)
	C-2	—
	St. C-4	—
	St. C-5	—
	St. C-6	—
1970		
Jan.	A-1	—
	A-2	—
	A-3	—
	A-4	—
	A-5	—
	C-1	—
	C-2	—
	St. C-4	—
	St. C-5	—
	St. C-6	—
Feb.	A-1	—
	A-2	—
	A-3	—
	A-4	—
	A-5	—
	C-1	—
	C-2	—
	C-3	—
	St. C-4	—
	St. C-5	—
	St. C-6	—
Mar.	A-1	—
	A-2	—
	A-3	—
	A-4	—
	A-5	—
	C-1	—
	C-2	—
	St. C-4	—
	St. C-5	—
	St. C-6	—
Apr.	A-1	—
	A-2	—
	A-3	—
	A-4	—
	A-5	—
	C-1	—
	C-2	—
	St. C-4	—
	St. C-5	—
	St. C-6	—
May	A-1	—
	A-2	—
	A-3	—
	A-4	—
	A-5	—
	C-1	—
	C-3	—
June	St. C-4	—
	St. C-5	—
	St. C-6	—

WATER

The results from the analyses of water samples for organochlorine insecticides are shown in Table 2. Sensitivities were 0.05 ppb for all insecticides in water except DDT and its isomers; the sensitivity for DDT and its isomers was 0.1 ppm. Concentrations below 0.1 ppb could not be accurately quantified and were reported as trace (<0.1 ppb). Eighty water samples were analyzed, and 3 (3.8%) showed the presence of organo-

chlorine insecticide residues. Heptachlor, endrin, and *p,p'*-DDT were the only organochlorine residues detected in the water samples. No correlation could be made between the insecticide residues detected and the chemical properties of the area.

These amounts represent only the insecticides dissolved in the water and do not include the suspended load of the water, i.e., suspended plankton, detritus, and silt. Therefore, the data are reasonable considering the low solubility of organochlorine insecticides in water, the adsorptive properties of sediments, and the ability of organisms to concentrate insecticide residues in their tissues.

SEDIMENT

The results from the analyses of sediment samples for organochlorine insecticides are shown in Table 3. The limit of detection was 0.1 ppb. Organochlorine insecticides were detected in only 3 (10.3%) of the 29 samples analyzed.

No correlation could be made between insecticide residues detected in the sediment and the chemical properties of the area. There was no relationship between insecticides reported in water samples and insecticides detected in the sediment; in each instance the residues detected in the water samples differed in type from the residues detected in the sediments. Heptachlor, endrin, and *p,p'*-DDT were detected in the water, but only dieldrin and *p,p'*-DDD were found in the sediments.

TABLE 3.—*Organochlorine residues in sediment, Aransas Bay, Texas—1969-70*
[— = not detected]

Collection Date	Sampling Location	Residues Detected in ppb (ng/g)
1969		
Sept.	A-1	dieldrin (0.7)
	A-2	—
	A-3	dieldrin (0.5)
	A-4	—
	A-5	—
Nov.	C-2	*p,p'*-DDD (24.6)
	St. C-4	—
	St. C-5	—
	St. C-6	—
1970		
Jan.	A-1	—
	A-2	—
	A-3	—
	A-4	—
	A-5	—
	C-1	—
	C-2	—
	St. C-4	—
	St. C-5	—
	St. C-6	—
May	A-1	—
	A-2	—
	A-3	—
	A-4	—
	A-5	—
	C-1	—
	C-3	—
June	St. C-4	—
	St. C-5	—
	St. C-6	—

ORGANISMS

The results from the analyses of organisms for organochlorine insecticides are shown in Table 4. The limit of detection was 0.1 ppb. Dieldrin, *p,p'*-DDE, *p,p'*-DDD, and *p,p'*-DDT were the only insecticides present in any of the organisms. The concentrations ranged from a trace (<0.1 ppb) to 66.5 ppb. The predominant insecticide residues detected in this investigation coincided with those found for this area by the Gulf Breeze Laboratory, and concentratons were similar.

TABLE 4.—*Organochlorine residues in organisms, Aransas Bay, Texas—1969-70*
[T = Trace = <0.1 ppb; — = not detected]

Collection Date	Type of Sample	Sampling Location	Residues Detected in ppb (ng/g)
1969			
July	2nd year, sea trout (muscle)	St. Chas. Bay	dieldrin (3.4) *p,p'*-DDE (25.5) *p,p'*-DDD (2.1) *p,p'*-DDT (5.5)
	2nd year, drum (whole)	St. Chas. Bay	dieldrin (9.7) *p,p'*-DDE (3.6) *p,p'*-DDT (2.1)
1970			
Jan.	oysters	A-1	*p,p'*-DDE (5.9) *p,p'*-DDD (T)
Mar.	blue crab (muscle)	Aransas Bay	*p,p'*-DDE (7.8) *p,p'*-DDD (T)
	blue crab (viscera)	Aransas Bay	—
	blue crab (eggs)	Aransas Bay	—
	silverside minnows	Aransas Bay	*p,p'*-DDE (33.3) *p,p'*-DDD (30.0) dieldrin (13.3)
	yellowtail croaker	Aransas Bay	*p,p'*-DDE (66.5) *p,p'*-DDD (60.0) *p,p'*-DDT (T)
	ribbon fish	Aransas Bay	*p,p'*-DDE (33.6) *p,p'*-DDD (41.0)
	brown shrimp	Aransas Bay	—
	yellowtail croaker	Aransas Bay	*p,p'*-DDE (25.0) *p,p'*-DDD (T)

Of particular interest is the lack of insecticide residues detected in shrimp and the low concentrations in the blue crabs and oysters. Butler (5) has postulated that the absence of insecticide residues in shrimp and other crustaceans is due to their high sensitivity to organochlorine insecticides, and that these animals are killed when the concentration of insecticides reaches a threshold value; this threshold may be low and thus, death may occur before these organisms have had time to metabolize the insecticide.

The low insecticide concentration in the oysters (*p,p'*-DDE, 5.9 ppb; *p,p'*-DDD, <0.1 ppb) is accounted for by the uncontaminated water and sediment at Station A-1. Although oysters can concentrate insecticides in their tissues when placed in contaminated water, they also have the ability, unlike many other organisms, to "flush" insecticide residues from their tissues when placed in uncontaminated water (*4*).

Butler (*6*) interpreted the presence of a high percentage of *p,p'*-DDT as indicating direct exposure to the insecticide and the presence of metabolites alone or at disproportionately high levels, as indicating that the residues have been transmitted through the food chain. A high percentage of *p,p'*-DDT was reported in the tissues of the sea trout and drum taken in July 1969. The first water samples analyzed (September 1969 and October 1969) indicated a trace of *p,p'*-DDT at Station A-1. No *p,p'*-DDT was found in any other water samples and only at one other time in any organism; this organism, the yellowtail croaker, (collected March 1970) had a trace of *p,p'*-DDT (<0.1 ppb) which constituted less than 0.1% of the total residues detected in the sample.

The small number of sediment and water samples in which organochlorine insecticides were present indicates that insecticide contamination of Aransas Bay is relatively low, and the presence of low organochlorine insecticide concentrations in the organisms may be attributed to transmission by the food chain.

Acknowledgment

Appreciation is expressed to the Texas Parks and Wildlife Department, Rockport, Tex., for their help in the collection of the samples and especially to Mr. T. L. Heffernan.

See Appendix for chemical names of compounds discussed in this paper.

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Total Mercury in Largemouth Bass (Micropterus salmoides) *in Ross Barnett Reservoir, Mississippi—1970 and 1971*

Luther A. Knight, Jr.[1] and Jack Herring[2]

ABSTRACT

Total mercury in 73 largemouth bass, Micropterus salmoides, *from Ross Barnett Reservoir, Mississippi, was measured by atomic absorption spectrophotometry. The fish analyzed were collected between November 1970 and October 1971 at intervals representing winter, spring, summer, and fall; specimens ranged in weight from 0.10 to 3.15 kg. Fish contained from <0.05 to 0.74 ppm total mercury; levels generally increased with weight of the fish.*

Introduction

Mercury compounds have been used for many years in agriculture, industry, and medicine, but only recently has there been much concern about the effects of mercurials on the aquatic environment. Previously, the assumption was generally accepted that metallic mercury settled to the bottom of a body of water and remained innocuous in the mud. However, research has shown that metallic mercury is converted through the action of microorganisms to methylmercury, and in this form is readily taken up and retained by living organisms (*7*). Most of this biological methylation of mercury is assumed to take place in the upper part of the bottom sediment (*10*). Eventually, by absorption and through the food chain, high levels of mercury may build up in carnivorous fishes such as the largemouth bass.

In 1970, concentrations of total mercury were found in fishes in Pickwick Lake, Mississippi, sufficient to warrant closing the lake to commercial fishing. With virtually little or no data available on the extent of contamination in Mississippi waters, a study was initiated to measure total mercury in Ross Barnett Reservoir. Preliminary results on total mercury content in largemouth bass (*Micropterus salmoides*) from the reservoir are given in this paper.

Materials and Methods

Fish were collected between November 1970 and October 1971 at intervals representing winter, spring, summer, and fall. All collections were made from the State Highway 43 area of Ross Barnett Reservoir except the September - October 1971 collections which included specimens from several other points in the reservoir as well (Fig. 1).

A flameless atomic absorption method described by Hatch and Ott (*8*) and Lee, Noble, and Randall (*11*) was used to measure total mercury. Hatch and Ott (*8*) stated that this method is accurate for determination of mercury at levels as low as 0.001 ppm. Bache, Gutenmann, and Lisk (*3*) stated that it is sensitive to at least 0.1 ppm mercury in fish. In the analyses reported here the sensitivity limit was determined to be 0.05 ppm and recovery was 87%. The results reported are corrected for recovery and for values obtained in analyzing control blanks.

Samples of flesh taken from the trunk region of the fish below the dorsal fin and above the lateral line were mechanically chopped and mixed in a Waring Blendor; 2- to 3-g subsamples were dissolved in 250-ml flasks with 5 ml each of concentrated nitric and sulfuric acids and then diluted with 50 ml of distilled water. After cooling the sample to room temperature, a 5-ml aliquot of 10% stannous sulfate was added as a reducing agent and the flask placed in the absorption system. Once maximum absorbance was obtained, the mercury vapor

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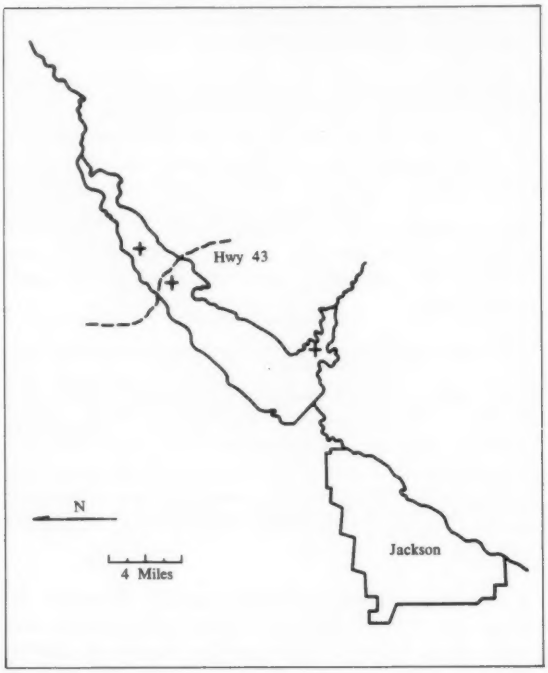
was vented to the hood and the flask removed. Standards were run in the same manner. Foaming caused by aeration of samples was abated with Dow Corning Antifoam A. A Beckman Atomic Absorption Spectrophotometer System equipped with a 10-inch Potentiometric Recorder was used for these analyses. Standards were obtained from Beckman Instruments, Inc., Fullerton, Calif.

FIGURE 1.—*Map of Ross Barnett Reservoir, Mississippi, showing locations (+) where fish were collected*



Results and Discussion

Residues of mercury in muscle tissues and length and weight of the fish sampled are given in Table 1. Mercury concentrations ranged from less than 0.05 to 0.74 ppm. In Table 2 average levels of total mercury in fish sampled are given according to weight groups. In general, average mercury levels increased with weight; fish weighing 0.5 kg or less contained less than 0.12 ppm mercury, and those weighing over 3 kg averaged 0.45 ppm. A study using largemouth bass from Ross Barnett Reservoir (*5*) has shown that length and weight are indicative of age. Further, Bache *et al.* (*3*) using tagged lake trout (*Salvelinus namaycush*), demonstrated positive correlation between age and accumulation of total mercury.

During the 12-month period in which fish were collected, temperatures ranged from about 8.0° to 29.0° C. Table 3 shows mercury residues in fish as a function of weight for each season of collection. All winter samples weighed less than 2 kg and with the exception of one sample containing 0.62 ppm generally had low levels of mercurial residues. All spring samples weighed 2 kg or less with the exception of two fish; residue levels in the smaller fish ranged from 0.10 to 0.48 ppm; the two larger fish contained 0.36 and 0.74 ppm. Samples collected in the summer showed less variation in the amount of mercury (range, 0.10 - 0.21 ppm); the two specimens weighing over 2 kg each contained 0.17 ppm. The fall collection contained a greater proportion of larger fish than collections from other seasons, and residues increased with weight more noticeably.

TABLE 1.—*Concentrations of total mercury (Hg) in largemouth bass by sample collection date—Ross Barnett Reservoir, 1970-71*

Length (cm)	Weight (kg)	Sex	Total Mercury Residue (ppm)
NOVEMBER 18—DECEMBER 30, 1970 (WINTER SAMPLES)			
22.23	0.14	—	<0.05
22.23	0.15	M	<0.05
22.23	0.15	—	<0.05
22.86	0.16	F	<0.05
22.86	0.17	—	<0.05
22.86	0.17	—	<0.05
22.86	0.17	F	<0.05
23.50	0.17	F	0.06
21.59	0.20	—	0.06
27.94	0.33	M	0.05
27.94	0.35	M	0.07
32.39	0.45	F	0.05
32.39	0.46	F	0.06
33.02	0.53	M	0.16
35.56	0.61	F	<0.05
37.47	0.82	F	<0.05
40.64	0.96	M	0.19
43.82	1.02	M	0.19
45.09	1.13	F	0.62
43.18	1.13	F	0.25
43.18	1.18	F	0.16
41.28	1.20	M	0.07
46.99	1.23	F	0.21
45.72	1.25	F	0.25
43.82	1.37	F	0.05
50.80	1.77	F	0.30
45.72	1.81	F	0.18
MARCH 29, 1971 (SPRING SAMPLES)			
20.32	0.13	F	0.10
22.23	0.15	M	0.38
23.50	0.24	F	0.36
25.40	0.24	M	0.10
26.04	0.29	F	0.24
30.48	0.36	F	0.10
30.48	0.39	M	0.23
30.48	0.46	M	0.17
32.00	0.48	M	0.18
32.39	0.48	M	0.11
30.48	0.52	M	0.24
33.02	0.59	F	0.27
37.08	0.70	M	0.14
38.10	0.70	M	0.20
37.08	0.74	M	0.48
38.10	0.79	M	0.27
46.99	1.35	F	0.27
57.79	2.61	F	0.36
58.42	3.15	F	0.74

TABLE 1.—*Concentrations of total mercury (Hg) in largemouth bass by sample collection date—Ross Barnett Reservoir, 1970-71*—Continued

Length (cm)	Weight (kg)	Sex	Total Mercury Residue (ppm)
JUNE 28—JULY 1, 1971 (SUMMER SAMPLES)			
19.69	0.10	F	0.11
20.32	0.12	F	0.13
21.16	0.14	M	0.12
—	0.14	F	0.10
25.40	0.19	M	0.12
29.85	0.40	F	0.14
31.75	0.42	F	0.15
33.02	0.50	M	0.16
38.10	0.68	F	0.16
39.12	0.72	M	0.14
39.37	0.74	F	0.13
41.91	1.04	F	0.16
46.36	1.37	F	0.21
59.06	2.66	F	0.17
54.61	3.09	F	0.17
SEPTEMBER 30 AND OCTOBER 1, 1971 (FALL SAMPLES)			
31.12	0.40	M	<0.05
36.83	0.77	M	<0.05
41.91	1.02	F	0.10
45.09	1.29	F	0.14
48.90	1.55	F	0.18
48.26	1.86	F	0.27
54.61	2.34	F	0.36
57.15	2.76	F	0.33
60.69	3.10	F	0.44
45.72	1.40	F	0.08
50.17	2.13	F	0.18
52.07	2.56	F	0.13

Total Number of Samples = 73

TABLE 2.—*Average concentrations of total mercury in Ross Barnett Reservoir largemouth bass according to weight groups*

Weight Group (kg)	Average Mercury Concentration in ppm	Number of Fish
0-0.50	0.12	31
0.51-1.00	0.18	15
1.01-1.50	0.20	14
1.51-2.00	0.23	4
2.01-2.50	0.27	2
2.51-3.00	0.25	4
3.01-3.15	0.45	3
Total		73

NOTE: In calculating average concentrations, those less than 0.05 ppm were computed at 0.05 ppm.

The ability of aquatic organisms to concentrate mercury above the level found in their environment is well known (*1,9*). However, the mechanism by which fish concentrate mercury is not fully understood. Rucker and Amend (*12*) found that mercury levels in muscle tissues from lake trout treated 1 hour per week for 12 weeks with 6.25% ethylmercury phosphate reached maximum concentrations of 4.4 ppm and returned to normal approximately 17 weeks following cessation of treatment. Studies have shown that fish absorb mercury compounds directly through their gills and through ingestion of a contaminated food (*7,12*).

TABLE 3.—*Total mercury in fish by weight of fish sampled*

Total Mercury (ppm)	Number of Fish per Weight Group			
	<1-1 kg	>1-2 kg	>2-3 kg	>3-4 kg
NOV. 18 & DEC. 30,1970 (WINTER SAMPLES)				
<0.05-0.1	15	2		
<0.1-0.2	2	3		
<0.2-0.3		4		
<0.3-0.4				
<0.4-0.5				
<0.5-0.6				
<0.6-0.7		1		
<0.7-0.8				
MARCH 29, 1971 (SPRING SAMPLES)				
<0.05-0.1	3			
<0.1-0.2	5			
<0.2-0.3	5	1		
<0.3-0.4	2		1	
<0.4-0.5	1			
<0.5-0.6				
<0.6-0.7				
<0.7-0.8				1
JUNE 28 & JULY 1, 1971 (SUMMER SAMPLES)				
<0.05-0.1	1			
<0.1-0.2	10	1	1	1
<0.2-0.3		1		
<0.3-0.4				
<0.4-0.5				
<0.5-0.6				
<0.6-0.7				
<0.7-0.8				
SEPT. 30 & OCT. 1, 1971 (FALL SAMPLES)				
<0.05-0.1	2	2		
<0.1-0.2		2	2	
<0.2-0.3		1		
<0.3-0.4			2	
<0.4-0.5				1
<0.5-0.6				
<0.6-0.7				
<0.7-0.8				

Feeding activities and levels of mercury in food chain organisms no doubt influence total mercury accumulation in largemouth bass. Dubets (*6*) and Schneidermeyer and Lewis (*13*) concluded that gizzard shad, when available, is the principal food of largemouth bass, and Applegate and Mullan (*2*) found that largemouth bass, after reaching 40 mm length, subsist principally on small shad which feed extensively on plankton and other small aquatic organisms. Barkley (*4,5,* and *personal communication*) found that on a percent by weight basis, shad comprised approximately 51.9% of the total fish population of Barnett Reservoir during the period from 1963 to 1970.

The sources of mercury available to the organisms in Ross Barnett Reservoir appear to be naturally occurring mercurials, items containing mercury which have been disposed of by the consumer public, and agricultural products; there are few industrial sources of mercury pollution in the drainage area of the Reservoir.

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A Survey of the Selenium Content of Fish From 49 New York State Waters

Irene S. Pakkala[1], Walter H. Gutenmann[1], Donald J. Lisk[1], George E. Burdick[2], and Earl J. Harris[3]

ABSTRACT

A survey was made of the selenium content of 438 fish of various species collected in 1969 from 49 New York State waters and a group of lake trout sampled in 1970 from Cayuga Lake only. Concentrations of selenium on a fresh-weight basis were usually below 1 ppm. There was little apparent correlation between selenium concentrations and species or sampling locations except that sturgeon from the Hudson River, lake trout from Lakes George and West Canada, whitefish from Raquette Lake, and several species from Lake Pleasant had consistently higher levels of selenium than other samples; all fish from Lakes Butterfield and Champlain and the Chenango and Salmon Rivers had consistently lower levels. No correlation was apparent between selenium levels and size or sex of fish. Selenium did not appear to be cumulative in lake trout of known age up to 12 years from Cayuga Lake.

Introduction

The contamination of fish by heavy metals and other toxic elements is presently of major concern. Selenium, although essential to man and animals in trace amounts, is very toxic at slightly higher levels. It is widely distributed in the lithosphere at a level of about 0.09 ppm (*5*) and is largely concentrated in sulfide materials. A major source of selenium as an air pollutant is the combustion of coal and oil, which have been found to contain concentrations up to 5 ppm (*6*) and 1.4 ppm (*11*), respectively. Selenium is also present as an impurity in fertilizers (*13*).

Although data on the levels of selenium in fish and aquatic organisms are scant, levels in zooplankton in Lake Michigan have been reported up to about .7 ppm (*4*); and another study has shown relatively higher levels, 1 to 2 ppm, in sea foods (*10*). The death of stocked game fish in a Colorado reservoir has been associated repeatedly with high levels of selenium in bottom sediments (*3*).

The present study was undertaken to determine the selenium concentrations in freshwater fish from 49 waters in New York State. A total of 438 fish were collected in 1969 by the New York State Conservation Department. In addition, a series of lake trout of increasing age up to 12 years were collected in an effort to determine if selenium was cumulative in these fish. Most of the samples, including the lake trout, were also analyzed for lead, and results of these analyses are reported in a previous issue of this Journal (*7*).

Sampling and Analytical Methods

The fish were netted, and species, sex, length, and weight recorded. All fish were decapitated and eviscerated. The remainder was chopped in a food chopper, mixed, and frozen in polyethylene bags prior to analysis. Selenium was determined by an adaptation of the method of Allaway and Cary (*1*) involving oxygen-flask combustion of 1 g of dried fish (*2*) and determination of the fluorescence of the selenium-2,3-diaminonaphthalene complex. Isolation of selenium by co-precipitation with arsenic (*1*) was found unnecessary and was thus omitted. The percent recoveries of selenium added to fish are given in Table 1. The method was sensitive to about 0.1 ppm of selenium in fish.

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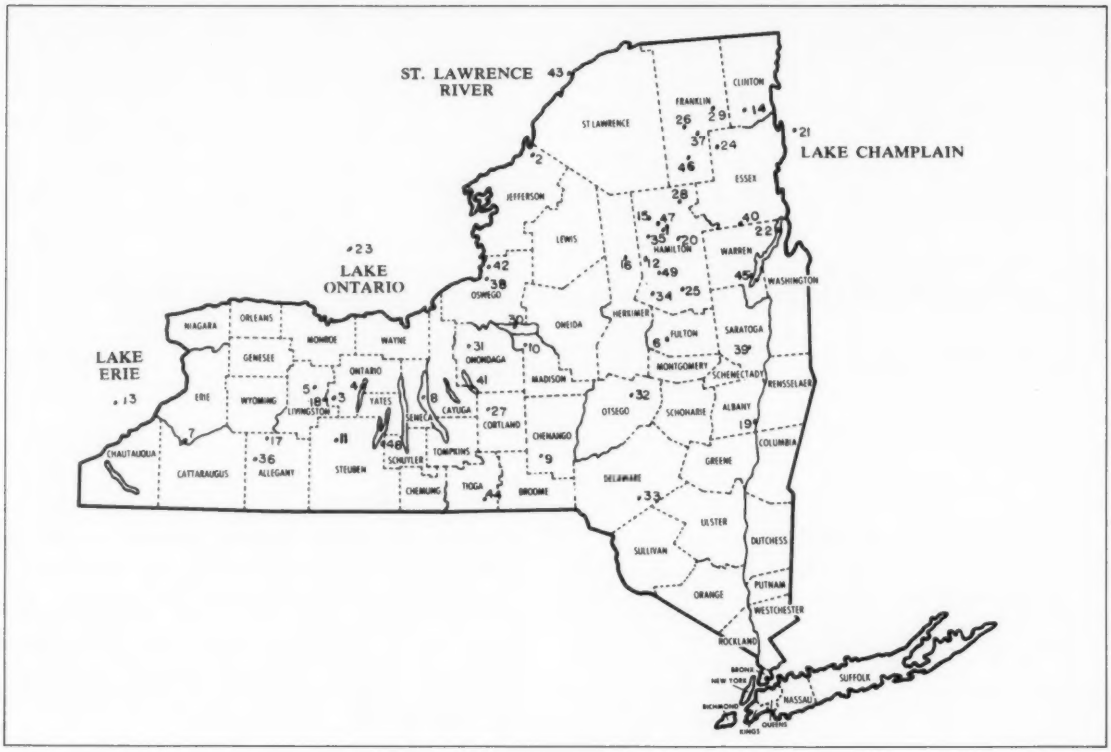
[3] N.Y. State Department of Environmental Conservation, Rome Pollution Laboratory, 8314 Fish Hatchery Rd., Rome, N.Y. 13440.

FIGURE 1.—*New York State map showing approximate locations of waters where fish were collected*



Results and Discussion

Table 2 lists the fish samples by their common and scientific names, and Fig. 1 designates numerically the approximate locations of the waters where fish were collected. The results of analysis of the fish for selenium, given in ppm on a fresh-weight basis and not corrected for percent recovery, are presented in Table 3. The fish species, sex, length, and weight are included to the extent that recording was complete.

Some general observations can be made concerning the selenium concentrations found (Table 3). All fish contained levels below 1 ppm, with the exception of four samples. The maximum concentration found was 1.50 ppm in a lake trout of unrecorded sex and size from Lake George. Levels were consistently higher in sturgeon from the Hudson River, lake trout from Lakes George and West Canada, whitefish from Raquette Lake, and various species of fish from Lake Pleasant. However, levels of selenium in fish from Lakes Erie and Ontario, generally considered quite polluted, were not notably high. Levels were consistently lower in fish from Lakes Butterfield and Champlain and the Chenango and Salmon Rivers. Selenium concentrations did not appear to be related to the size, sex, or species of fish, and, with the exception of the observations noted above, levels did not appear to be related to location of waters from which fish were collected. Other observations might have been apparent had more fish from specific waters been analyzed.

Virtually no data are available on the soil and mineral content of selenium in New York and, therefore, attempts to correlate selenium concentrations in fish with natural background levels were unsuccessful.

Table 4 lists the selenium concentrations in lake trout, of increasing age up to 12 years, from Cayuga Lake. The ages were accurately known since fingerling lake trout are stocked there annually and marked with the date of stocking. Based on these limited data, selenium did not seem to be cumulative in this species.

The levels of selenium found in fish in this study are in the same range as those present in many other common food classes (*10*) and are not considered significant. In this regard, it is interesting to note that selenium has been reported to be markedly protective against the toxic effects of mercury and cadmium (*8,9,12*).

Acknowledgment

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TABLE 1.—*Percent recovery of selenium from fish*

Species	Selenium Added (ppm)	Percent Recovery
Brown catfish	0.2	105.85
	0.5	68.98
	1.0	84.94
Lake trout	0.5	100
	1.0	97
Largemouth bass	0.2	75.60
	0.5	86.88
	1.0	85.69
Northern pike	0.2	80.90
	0.5	80.68
	1.0	89.89

NOTE: Sensitivity level = 0.1 ppm.

TABLE 2.—*Common and scientific names of fish analyzed in this study*

Common Name	Scientific Name
Black crappie	*Pomoxis nigromaculatus*
Bowfin	*Amia calva*
Brook trout (Speckled trout)	*Salvelinus fontinalis*
Brown trout	*Salmo trutta*
Brown catfish (bullhead) and Channel catfish	*Ictalurus sp.*
Burbot	*Lota lota*
Carp	*Cyprinus carpio*
Chain pickerel	*Esox niger*
Cisco	*Coregonus artedii*
Coho salmon	*Oncorhynchus kisutch*
Freshwater drum	*Aplodinotus grunniens*
Gizzard shad	*Dorosoma cepedianum*
Goldfish	*Carassius auratus*
Lake trout	*Salvelinus namaycush*
Lake whitefish	*Coregonus clupeaformis*
Largemouth bass	*Micropterus salmoides*
Muskellunge	*Esox masquinongy*
Northern pike	*Esox lucius*
Perch	*Perca sp.*
Rainbow trout	*Salmo gairdneri*
Rock bass	*Ambloplites rupestris*
Smallmouth bass	*Micropterus dolomieui*
Splake (Brook and Lake trout cross)	
Striped bass	*Morone saxatilis*
Sturgeon, Atlantic sturgeon, and Shortnose sturgeon	*Acipenser sp.*
Walleye pike	*Stizostedion vitreum vitreum*
White bass (Silver bass)	*Roccus chrysops*
White sucker (Sucker)	*Catostomus commersoni*

TABLE 3.—*Residues of total selenium in fish from New York State waters in 1969*

Species [1]	Tag No.	Sex	Length (cm)	Weight (kg)	Selenium Residue Level (ppm)
BLUE MOUNTAIN LAKE—NO. 1 [2]					
Bullhead catfish	6R6527	F	28.4	.379	0.26
	6R6528	M	28.9	.377	0.48
	6R6529	F	27.9	.378	0.38
Rainbow trout	6R6522	F	33.0	.393	0.65
	6R6521	M	33.8	.173	0.57
Smallmouth bass	6R6526	M	30.5	.337	0.64
	6R6524	M	28.2	.275	0.50
BUTTERFIELD LAKE—NO. 2 [2]					
Bowfin	22BL-4	M	55.9	1.702	0.15
Bullhead catfish	24BL-4	F	32.2	.669	0.10
	23BL-4	F	33.0	.685	0.11
	25BL-4	F	32.0	.680	0.10
Largemouth bass	19BL-4	F	34.8	.657	0.21
	21BL-4	M	34.5	.626	0.18
	20BL-4	F	33.0	.573	0.11
Northern pike	9BL-4	—	—	—	0.15
	8BL-4	—	—	—	0.15
	10BL-4	—	—	—	0.21
	7BL-4	—	—	—	0.15
Rock bass	14BL-4	M	22.9	.263	0.22
	15BL-4	M	22.2	.277	0.15
Smallmouth bass	11BL-4	—	—	—	0.22
	12BL-4	—	—	—	0.21
Sucker	28BL-4	M	38.6	.916	0.18
Walleye pike	26BL-4	M	55.9	1.753	0.17
	29BL-4	M	47.3	1.071	0.20
	27BL-4	M	48.7	1.268	0.19
CANADICE LAKE—NO. 3 [2]					
Bullhead catfish	J6220	—	35.0	.574	0.12
Lake trout	J6293	M	47.8	1.246	0.42
	J6295	M	76.8	4.855	0.44
Chain pickerel	J6228	F	46.0	.699	0.32
	J6222	F	40.7	.471	0.24
Rainbow trout	J6225	—	29.2	.284	0.36
	J6296	M	30.5	.370	0.31
	J6297	M	32.5	.430	0.27
Rock bass	J6212	M	21.8	.248	0.39
Smallmouth bass	J6210	M	36.1	.751	0.41
	J6211	M	28.4	.386	0.53
	J6291	M	29.2	.409	0.50
	J6290	F	33.6	.603	0.48
	J6292	F	29.2	.378	0.37
CANANDAIGUA LAKE—NO. 4 [2]					
Brown trout	J6288	M	56.7	2.595	0.39
Lake trout	J6287	M	55.2	1.821	0.36
	J6286	M	52.6	1.545	0.41
	J6231	[3]M-Imm.	44.4	.722	0.55
Largemouth bass	J6167	M	28.7	.370	0.18
Smallmouth bass	J6162	F	28.7	.359	0.63
CONESUS LAKE—NO. 5 [2]					
Black crappie	J6102	—	—	—	0.20
CAROGA LAKE—NO. 6 [2]					
Smallmouth bass	6R6192	F	40.9	1.295	0.42

TABLE 3.—*Residues of total selenium in fish from New York State waters in 1969*—Continued

Species [1]	Tag No.	Sex	Length (cm)	Weight (kg)	Selenium Residue Level (ppm)
CATTARAUGUS CREEK—NO. 7 [2]					
Coho salmon	2F1201	M	41.6	.683	0.40
	2F1202	M	40.2	.588	0.28
	2F1203	M	37.9	.588	0.45
	2F1204	F	54.4	1.77	0.26
	2F1205	F	56.8	2.09	0.32
	2F1206	F	61.0	2.19	0.24
	2F1207	F	60.2	2.23	0.24
	2F1208	F	63.0	3.04	0.27
	2F1209	F	64.0	3.00	0.25
CAYUGA LAKE—NO. 8 [2]					
Largemouth bass	3-CL-2	—	15.25	—	0.25
	3-CL-1	M	31.7	.621	0.29
Chain pickerel	CAY-1	—	—	—	0.23
	CAY-2	—	—	—	0.22
	CAY-3	—	—	—	0.34
CHENANGO RIVER—NO. 9 [2]					
Smallmouth bass	3-CHR-1	F	22.2	.168	0.28
	3-CHR-3	F	27.9	.343	0.24
	3-CHR-2	[3]—Imm.	24.8	.198	0.23
	3-CHR-4	M	28.6	.295	0.27
	3-ChR.Ch.-8	F	30.5	.436	0.21
	3-CHR-5	M	34.9	.550	0.23
CHITTENANGO CREEK—NO. 10 [2]					
Brown trout	3-CH-1	F	43.2	1.072	0.26
	3-CH-2	F	31.7	.447	0.24
	3-CH-3	F	23.4	.151	0.27
COHOCTON RIVER—NO. 11 [2]					
Brown trout	2CohR1	M	27.6	.256	0.40
	2CohR2	F	26.7	.251	0.32
	2CohR3	M	25.4	.194	0.33
EIGHTH LAKE—NO. 12 [2]					
Rainbow trout	6R6493	—	—	—	0.41
LAKE ERIE—NO. 13 [2]					
Burbot	2-LAER-6	—	58.4	1.980	0.33
Coho salmon	2-LE-Hg-7	F	41.3	.665	0.37
	2-LE-Hg-6	F	44.5	.764	0.33
	LE-CS 1-9-4-69	M	56.2	2.110	0.35
	2-LE-Hg-8	M	48.3	1.147	0.45
(Sunset Bay)	LE-CS-9	F	57.5	2.671	0.35
Do.	LE-CS-8	F	55.0	2.392	0.34
Do.	LE-CS-6	[3]M-Imm.	35.0	.493	0.36
Do.	LE-CS-7	F	56.8	2.520	0.31
Freshwater drum	LE-FWD-1	M	45.0	1.005	0.81
	2-LE-Hg-3	F	40.0	.733	0.45
	2-LE-Hg-42	F	36.8	.670	0.42
	2-LE-Hg-37	M	36.2	.650	0.35
	2-LE-Hg-47	F	48.3	1.385	0.28
	2-LE-Hg-38	M	37.5	.715	0.48
	2-LE-Hg-45	M	34.3	.570	0.62
	2-LE-Hg-43	M	35.6	.610	0.43
	2-LE-Hg-41	F	37.5	.750	0.59
	2-LE-Hg-46	M	36.2	.565	0.42
	2-LE-Hg-40	F	32.4	.415	0.38
	2-LE-Hg-44	M	36.8	.610	0.41
	2-LE-Hg-39	—	—	—	0.42
	2-LE-Hg-48	M	48.9	1.395	0.24
	2-LE-1	—	43.2	.695	0.29
	2-LE-3	F	30.7	.270	0.43
	2-LE-Hg-2	F	36.2	.612	0.32
	2-LE-2	F	50.8	1.750	0.37
	2-LE-Hg-1	M	38.1	.615	0.36
	LE-FWD-2	M	29.2	.364	0.54

TABLE 3.—*Residues of total selenium in fish from New York State waters in 1969*—Continued

Species [1]	Tag No.	Sex	Length (cm)	Weight (kg)	Selenium Residue Level (ppm)
LAKE ERIE—NO. 13 [2]—Continued					
Gizzard shad	2-LE-4	Composite of 31 fish			0.42
Lake trout	2-LEL-2	—	—	—	0.33
Rock bass	2-LE-Hg-31	M	27.3	.358	0.31
Silver bass	2-LE-Hg-51	F	37.5	.680	0.52
	2-LE-Hg-68	F	36.2	.487	0.37
	2-LE-Hg-71	F	35.6	.487	0.43
	2-LE-Hg-53	F	36.2	.615	0.32
	2-LE-Hg-70	M	34.9	.503	0.46
	2-LE-Hg-52	F	36.2	.625	0.35
	2-LE-Hg-72	F	40.0	.840	0.41
	2-LE-Hg-69	M	34.9	.425	0.51
Smallmouth bass	LE-SMB-1	M	43.5	.932	0.40
	2-LE-Hg-54	F	40.7	.860	0.30
	2-LE-Hg	—	—	—	0.31
	2-LE-Hg-50	F	34.9	.645	0.29
	2-LE-Hg-60	F	39.4	.649	0.57
	2-LE-Hg-62	F	45.7	1.300	0.43
	2-LE-Hg-58	M	33.0	.360	0.37
	2-LE-Hg-56	F	35.6	.682	0.38
	2-LE-Hg-59	F	37.5	.741	0.54
	2-LE-Hg-57	M	40.0	.695	0.26
Sucker	2-LAER-5	—	50.8	1.474	0.59
	2-LE-Hg-13	F	51.5	1.530	0.47
	2-LAER-4	M	30.5	.334	0.34
	2-LE-Hg-12	F	48.9	1.296	0.37
	2-LE-Hg-65	F	44.5	1.150	0.51
	2-LE-Hg-63	M	46.4	1.190	0.71
	2-LAER-3	M	48.3	1.097	0.21
	2-LE-Hg-11	M	45.7	1.040	0.36
	2-LE-Hg-64	F	46.4	1.260	0.44
	2-LE-Hg-67	F	43.2	1.120	0.40
	2-LE-Hg-66	F	49.5	1.150	0.33
Walleye pike	2-LE-Hg-23	F	69.3	4.210	0.30
	2-LE-Hg-25	F	64.7	2.920	0.20
	2-LE-Hg-32	F	63.5	3.375	0.25
	2-LE-Hg-4	F	63.5	2.310	0.37
	2-LE-Hg-33	M	73.7	5.070	0.40
	2-LE-Hg-35	F	67.4	3.800	0.23
	2-LE-Hg-36	F	62.3	3.380	0.33
	2-LE-Hg-21	F	68.5	4.219	0.25
Yellow perch	2-LE-Hg-16	F	31.7	.515	0.31
	2-LE-Hg-29	F	33.0	.609	0.37
	2-LE-Hg-30	F	27.9	.485	0.27
	2-LE-Hg-17	F	31.1	.516	0.35
	2-LE-Hg-28	F	34.3	.697	0.28
	2-LE-Hg-26	F	32.4	.612	0.27
	2-LE-Hg-20	F	33.3	.605	0.32
	2-LE-Hg-19	F	36.2	.735	0.35
	2-LE-Hg-18	F	34.9	.694	0.34
	2-LE-Hg-27	F	30.5	.450	0.36
FERN LAKE—NO. 14 [2]					
Rock bass	5-FLK-3	M	26.2	.426	0.25
FORKED LAKE—NO. 15 [2]					
Brook trout	6R6641	—	43.7	.945	0.40
Lake trout	6R6642	M	53.3	.854	0.33
Smallmouth bass	6R6648	F	37.1	.950	0.39
Speckled trout	6R6640	F	35.6	.645	0.57
FOURTH LAKE—NO. 16 [2]					
Lake trout	4-4L-12	F	55.9	1.872	0.39
	4-4L-10	M	37.8	.497	0.37
	4-4L-11	—	—	—	0.33
Sucker	4-4L-23	M	29.2	.301	0.27
	4-4L-22	F	30.2	.336	0.40
	4-4L-24	M	35.6	.554	0.72

TABLE 3.—*Residues of total selenium in fish from New York State waters in 1969*—Continued

Species [1]	Tag No.	Sex	Length (cm)	Weight (kg)	Selenium Residue Level (ppm)
GENESSEE RIVER—NO. 17 [2]					
Smallmouth bass	J-6267	—	—	—	0.35
HEMLOCK LAKE—NO. 18 [2]					
Bullhead catfish	J6203	M	36.1	.739	0.1
	J6202	—	28.2	.341	0.18
Chain pickerel	J6198	—	31.7	.229	0.44
	J6199	M	45.7	.754	0.56
	J6197	—	50.8	1.245	0.38
Lake trout	J6299	M	67.5	3.987	0.42
	J6298	F	77.8	4.270	0.39
	J6300	M	75.5	4.580	0.47
Largemouth bass	J6229	M	26.2	.344	0.19
	J6201	—	28.4	.468	0.42
Rainbow trout	J6200	M	38.1	.688	0.45
HUDSON RIVER—NO. 19 [2]					
Atlantic sturgeon	J2067	[3]—Imm.	46.0	.410	0.69
	J2065	[3]F-Imm.	62.0	1.285	0.84
	J2064	M	98.0	5.076	0.80
	J2066	[3]M-Imm.	64.0	1.075	0.26
	J2068	[3]—Imm.	64.5	1.348	0.88
Goldfish	6R6667	—	—	—	0.18
	6R6666	—	—	—	0.17
	6R6665	—	—	—	0.14
Shortnose sturgeon	J2076	[3]—Imm.	47.3	.636	0.37
Striped bass	J2070	M	57.6	2.256	0.25
	J2071	M	53.4	2.042	0.37
	J2072	F	41.1	.780	0.24
	J2069	M	54.4	2.072	0.65
	8HUDR2	M	40.6	.800	0.38
	8HUDR3	M	43.7	.985	0.30
	8HUDR1	M	40.2	.850	0.45
INDIAN LAKE—NO. 20 [2]					
Northern pike	6R6554	M	73.0	2.735	0.27
	6R6142	M	64.3	1.433	0.35
Walleye pike	6R6553	M	70.6	2.562	0.27
	6R6144	M	29.7	.380	0.33
Whitefish	6R6139	F	38.6	.595	0.43
	6R6140	M	43.2	.789	0.32
	6R6141	M	45.3	.641	0.71
	6R6143	M	46.0	.873	0.24
LAKE CHAMPLAIN—NO. 21 [2]					
Channel catfish	5-LCCC-1	M	53.4	1.630	0.14
	5-LCCC-2	F	76.1	6.200	0.11
	5-LCCC-3	—	61.0	2.630	0.12
	6R6592	F	52.0	1.700	0.12
(South Bay)	6R6591	M	47.8	1.130	0.15
Do.	6R6593	M	60.5	2.760	0.14
Freshwater drum					
(South Bay)	6R6588	F	31.7	.306	0.24
Do.	6R6590	F	48.8	1.165	0.11
Do.	6R6589	M	36.8	.423	0.22
Walleye pike					
(South Bay)	6R6587	M	64.8	3.070	0.24
Do.	6R6585	F	40.2	.566	0.23
Do.	6R6586	M	54.9	1.455	0.26
LAKE GEORGE—NO. 22 [2]					
Black crappie	6R6676	F	20.3	.109	0.38
	6R6675	F	31.2	.584	0.35
	6R6452	F	28.7	.352	0.47

TABLE 3.—*Residues of total selenium in fish from New York State waters in 1969*—Continued

Species [1]	Tag No.	Sex	Length (cm)	Weight (kg)	Selenium Residue Level (ppm)
LAKE GEORGE—NO. 22 [2]—Continued					
Brown bullhead	6R6680	F	32.8	.686	0.28
	6R6454	F	32.8	.508	0.29
Chain pickerel	6R6450	M	35.6	.258	0.30
Lake trout	6R6672	F	71.0	3.570	0.46
	6R573	—	—	—	1.50
	6R6674	M	68.8	3.125	0.78
	6R6673	F	70.1	3.241	0.68
Largemouth bass	6R6441	M	41.1	.973	0.51
	6R6453	F	36.3	.743	0.40
	6R6440	M	40.4	.789	0.57
Northern pike	6R6677	F	64.0	1.820	0.38
	6R6671	F	59.5	1.780	0.41
	6R6404	F	75.0	3.180	0.44
	6R6407	F	64.7	2.950	0.35
	6R6406	M	60.5	3.000	0.45
Rainbow trout	6R6681	M	55.3	2.440	0.55
	6R6682	M	50.8	1.690	0.53
	6R639	F	41.9	.692	0.52
	6R6425	F	58.6	2.410	0.63
Smallmouth bass	6R6449	F	42.4	1.235	0.49
White sucker	6R6433	F	53.0	1.139	0.59
	6R6434	M	46.3	.872	0.60
	6R6432	M	47.3	.906	0.48
LAKE ONTARIO—NO. 23 [2]					
Black crappie	2-LO-4	—	—	—	0.26
Carp	J6067	—	27.4	.419	0.21
Coho salmon	69-LO-CS1	F	57.7	2.555	0.34
	20-LO-4	M	58.5	2.640	0.30
	21-LO-4	F	48.3	.965	0.29
	22-LO-4	M	50.0	1.400	0.30
	23-LO-4	F	55.9	2.180	0.39
	73-30-69	F	51.3	2.060	0.39
(False Duck Is.)	73-29-69	F	55.8	2.040	0.38
(Outlet Beach)	73-49-69	F	63.0	3.110	0.43
(Amherst Bar)	73-18-69	M	60.1	3.190	0.47
(Pt. Petre)	73-27-69	F	56.1	2.090	0.31
(Perch Cove)	73-23-69	M	60.4	2.090	0.50
(Shelter Valley Mouth)	73-47-69	F	64.0	2.890	0.44
(Pennicon Reef)	73-16-69	F	56.8	2.460	0.37
Do.	73-13-69	M	50.3	1.740	0.29
	73-13-69 (testes)	M	—	—	0.25
Do.	73-17-69	M	64.8	3.780	0.38
(Wellington Beach)	73-51-69	M	60.5	2.120	0.46
Do.	73-44-69	M	64.8	3.010	0.40
Do.	73-53-69	M	59.9	2.100	0.68
Do.	73-52-69	F	66.0	3.290	0.44
Do.	73-48-69	F	64.8	3.010	0.47
Do.	73-50-69	M	54.0	1.620	0.49
Rock bass	J6038	F	22.8	.268	0.31
	J6097	F	24.1	.288	0.35
	J6098	M	25.5	.436	0.31
	J6085	F	22.8	.301	0.31
Smallmouth bass	J-6165	—	—	—	0.28
	9-LO-4	F	37.6	.826	0.43
	16-LO-4	M	40.7	1.014	0.35
(Carleton Is.)	4CI-1	M	28.9	.320	0.37
Do.	4CI-2	M	27.4	.258	0.39
Do.	4CI-3	M	34.0	.573	0.38

TABLE 3.—*Residues of total selenium in fish from New York State waters in 1969*—Continued

Species [1]	Tag No.	Sex	Length (cm)	Weight (kg)	Selenium Residue Level (ppm)
LAKE ONTRARIO—NO. 23 [2]—Continued					
Smallmouth bass (Cont'd)					
(Pt. Peninsula)	4RA2637	F	37.4	.855	0.30
Do.	4RA2635	F	33.0	.558	0.37
Do.	4RA2632	—	—	—	0.32
Do.	4RA2638	F	33.6	.644	0.37
Do.	4RA2646	F	30.5	.533	0.38
Do.	4RA2644	M	38.1	.892	0.40
Do.	4RA2650	M	27.9	.320	0.49
Splake	17-LO-4	[3]—Imm.	34.8	.523	0.48
Sucker	15-LO-4	M	39.1	.657	0.24
	14-LO-4	F	39.1	.703	0.20
White bass	J6027	F	25.5	.272	0.24
	J6030	—	24.9	.207	0.50
LAKE PLACID—NO. 24 [2]					
Brook trout	LP-NE-2	F	30.5	.360	1.10
Lake trout	5-NELP-2	—	41.5	.726	0.77
(North End)	5-NELP-1	M	31.7	.299	0.80
Northern pike	3LP-5	M	76.6	3.870	0.40
	1LP-5	M	57.4	1.408	0.34
	2LP-5	M	64.0	1.642	0.34
Rainbow trout	5-LKP-1	F	25.4	.225	0.29
	5-LKP-3	F	40.9	.715	0.51
	5-LKP-2	F	42.1	.779	0.55
Smallmouth bass	5-LKP-4	M	29.2	.336	0.76
	5-LKP-5	F	46.3	1.420	0.45
White sucker	5-LP-WS-1	—	43.7	1.440	0.37
	5-LP-WS-3	—	30.5	.417	0.65
Yellow perch	5-LP-5	F	31.7	.435	0.77
LAKE PLEASANT—NO. 25 [2]					
Brown bullhead	6R6118	M	34.3	.681	0.38
	6R6119	F	34.8	.673	0.31
Bullhead catfish	6R6120	M	25.5	.309	0.24
Chain pickerel	6R6126	F	35.0	.255	0.40
	6R6127	F	38.4	.392	0.29
	6R6128	F	39.2	.405	0.25
Largemouth bass	6R6131	F	35.0	.622	0.24
Rainbow trout	6R6129	F	42.7	1.068	1.05
Rock bass	6R6124	F	26.4	.461	0.98
	6R6123	F	25.4	.325	0.78
Smallmouth bass	6R6125	—	—	—	0.84
	6R6132	F	32.7	.484	0.40
	6R6130	F	49.3	1.775	0.74
Whitefish	6R6117	M	53.8	2.153	0.62
	6R6116	—	61.0	2.035	0.82
Yellow perch	6R6135	M	30.0	.438	0.86
	6R6133	M	27.4	.335	0.92
LITTLE GREEN POND—NO. 26 [2]					
Rainbow trout	5-lgp-8	F	34.1	.392	0.67
	5-lgp-7	M	32.0	.370	0.37
	5-lgp-9	F	38.6	.608	0.44
Whitefish	5-lgp-1	F	53.4	1.805	0.97
	5-lgp-2	F	57.1	2.041	0.74
	5-lgp-3	M	50.8	2.050	0.49

TABLE 3.—*Residues of total selenium in fish from New York State waters in 1969*—Continued

Species [1]	Tag No.	Sex	Length (cm)	Weight (kg)	Selenium Residue Level (ppm)
LITTLE YORK LAKE—NO. 27 [2]					
Brown trout	LY-3-1	M	34.3	.456	0.23
Bullhead catfish	LY-3-3	F	31.7	.414	0.10
Largemouth bass	LY-3-2	M	33.2	.720	0.15
LONG LAKE—NO. 28 [2]					
Largemouth bass	6R6546	F	43.2	1.128	0.45
Northern pike	6R6541	F	53.8	1.167	0.40
	6R6540	M	71.4	2.305	0.54
	6R6542	M	61.0	1.335	0.55
Smallmouth bass	6R6545	F	40.9	.767	0.54
	6R6544	F	35.6	.672	0.51
LOON LAKE—NO. 29 [2]					
Smallmouth bass	6R6414	—	—	[4].201	0.15
Walleye pike	6R6409	F	—	[4].190	0.22
	6R6410	F	—	[4].187	0.19
	6R6408	F	—	[4].232	0.24
ONEIDA LAKE— NO. 30 [2]					
Walleye pike	3-OneL-1	F	59.6	2.060	0.23
	3-OneL-2	F	48.3	1.030	0.17
ONONDAGA CREEK—NO. 31 [2]					
Brown trout	OS-P154-4-1	F	32.2	.386	0.43
	OS-P154-4-2	—	29.5	.314	0.33
	OS-P154-4-3	—	24.1	.181	0.44
OTSEGO LAKE—NO. 32 [2]					
Cisco	7-OTSL-1	—	25.9	.161	0.30
	7-OTSL-2	M	34.3	.365	0.33
	7-OTSL-3	F	36.1	.488	0.36
Lake trout	7-OTSL-6	—	—	—	0.34
	7-OTSL-5	—	—	—	0.33
	7-OTSL-4	—	—	—	0.33
Whitefish	7-OTSL-9	—	—	—	0.33
	7-OTSL-7	—	—	—	0.56
	7-OTSL-8	—	—	—	0.42
PEPACTON RESERVOIR—NO. 33 [2]					
Brown trout	1PR-7	F	64.8	3.115	0.40
	3PR-7	F	51.5	1.880	0.35
	2PR-7	F	41.2	.953	0.37
PISECO LAKE—NO. 34 [2]					
Brown bullhead	6R6105	M	31.7	.425	0.14
Smallmouth bass	6R6104	F	35.3	.571	0.43
Whitefish	6R6103	F	38.1	.431	0.52
	6R6102	F	31.7	.295	0.48
	6R6101	M	34.8	.328	0.43
Yellow perch	6R6111	M	29.2	.262	0.65
RAQUETTE LAKE—NO. 35 [2]					
Lake trout	6R6523	—	—	—	0.97
Smallmouth bass	6R6207	F	28.9	.287	0.37
	6R6206	F	27.6	.287	0.31
	6R6208	F	27.4	.251	0.35

TABLE 3.—*Residues of total selenium in fish from New York State waters in 1969*—Continued

Species [1]	Tag No.	Sex	Length (cm)	Weight (kg)	Selenium Residue Level (ppm)
RAQUETTE LAKE—NO. 35[2]—Continued					
Whitefish	6R6245	[3]—Imm.	33.0	.337	0.59
	6R6244	F	34.8	.453	0.76
	6R6625	F	31.7	.349	0.93
	6R6243	F	35.6	.497	0.75
	6R6627	M	34.0	.388	0.52
	6R6626	M	35.3	.439	0.84
RUSHFORD LAKE—NO. 36 [2]					
Lake trout	2RL-1	F	75.2	4.195	0.23
SARANAC LAKE—NO. 37 [2]					
Smallmouth bass	5SL-5	F	33.2	.684	0.30
SALMON RIVER—NO. 38 [2]					
Coho salmon	69-SR-CS-6	M	61.5	1.910	0.29
(Pulaski, N.Y.)	69-SR-CS-4	F	56.8	1.983	0.27
	69-SR-CS-2	F	63.0	2.072	0.31
	69-SR-CS-10	M	61.8	1.920	0.34
	69-SR-CS-10 (testes)	M	—	—	0.18
	69-SR-CS-7	M	60.5	2.046	0.31
	69-SR-CS-8	M	57.3	1.984	0.44
	SR-CS-1	F	48.8	1.036	0.27
	69-SR-CS-5	F	53.9	1.661	0.31
	69-SR-CS-11	F	61.8	1.840	0.29
	CSJ-SR-2	M	35.6	.620	0.31
	CSJ-SR-1	M	40.7	.663	0.31
	CSJ-SR-3	M	38.1	.544	0.30
	69-SR-CS-9	F	61.8	1.735	0.35
	SRCSJ-70-1	[3]—Imm.	33.0	.444	0.35
	SRCSJ-70-2	do.	30.5	.280	0.34
	SRCSJ-70-3	do.	34.9	.520	0.33
	SRCSJ-70-4	do.	32.4	.400	0.27
	SRCSJ-70-5	do.	30.9	.326	0.34
	SRCSJ-70-6	do.	30.9	.327	0.36
SARATOGA LAKE—NO. 39 [2]					
Walleye pike	6R6403	F	—	.357	0.22
SCHROON LAKE— NO. 40 [2]					
Lake trout	1	—	—	—	0.27
SKANEATELES LAKE—NO. 41 [2]					
Rainbow trout	3-SKL-1	F	50.8	1.235	0.56
	3-SKL-3	F	38.1	.644	0.71
SPRING BROOK—NO. 42 [2]					
Coho salmon	5M381	M	54.4	1.710	0.36
	4RA2991	—	56.8	2.160	0.37
	4RA3601	—	61.5	2.680	0.38
ST. LAWRENCE RIVER—NO. 43 [2]					
Brown bullhead	9SL-5	—	—	—	0.14
Muskellunge	101SL-4	M	77.5	3.832	0.27
Smallmouth bass	1STL-4	M	29.5	.398	0.34
	6STL-4	M	26.8	.263	0.41
	2STL-4	M	26.9	.280	0.41
	6SL-5	—	—	—	0.35
	3STL-4	[3]—Imm.	23.9	.242	0.43
	4STL-4	F	30.0	.339	0.34
	5STL-4	M	29.5	.344	0.39
Sturgeon	G429	—	102.5	6.830	0.33
	G435	—	89.6	4.540	0.34
	G430	—	—	3.630	0.23
	G432	—	97.3	5.690	0.33
	G434	—	100.8	6.590	0.25
	G431	—	103.5	8.200	0.29
	G433	—	96.3	5.910	0.33

TABLE 3.—*Residues of total selenium in fish from New York State waters in 1969*—Continued

Species [1]	Tag No.	Sex	Length (cm)	Weight (kg)	Selenium Residue Level (ppm)
ST. LAWRENCE RIVER—NO. 43[2]—Continued					
Sturgeon	1-SL-54	—	—	—	0.38
(Cont'd)	3-SL-56	—	—	—	0.35
(Massena)	2-SL-56	—	—	—	0.46
Do.	4-SLS-6	M	95.5	7.210	0.31
Walleye pike	1-STL-Hg-4	—	—	—	0.31
(Massena)	4-MAS-1	—	—	—	0.30
Do.	4-MAS-2	F	45.3	1.060	0.31
	4-MAS-3	F	65.5	3.527	0.30
SUSQUEHANNA RIVER—NO. 44 [2]					
Walleye pike	3-SQR-1	—	—	—	0.19
	3-Susq.R.-6	M	22.8	.125	0.20
	3-Susq.R.-7	M	33.0	.305	0.24
Yellow perch	3-Susq.R.-5	M	27.9	.268	0.18
	3-Susq.R.-4	M	24.8	.166	0.21
	3-Susq.R.-3	—	—	—	0.19
TROUT LAKE—NO. 45 [2]					
Chain pickerel					
(West Shore)	6R6456	F	62.0	1.570	0.20
Do.	6R6457	M	46.3	.785	0.27
	6R6458	F	49.5	.918	0.24
Smallmouth bass	6R6459	F	43.7	1.197	0.30
UPPER SARANAC LAKE—NO. 46 [2]					
Largemouth bass	5-US-2	F	34.8	.904	0.34
	5-US-1	F	31.2	.432	0.32
UTOWANA LAKE—NO. 47 [2]					
Smallmouth bass	6R6504	F	42.9	.949	0.53
	6R6505	M	42.5	1.030	0.60
Rainbow trout	6R6506	M	43.0	.844	0.40
	6R6508	F	41.5	.775	0.40
WANETA LAKE—NO. 48 [2]					
Muskellunge	1ML-2	—	—	—	0.15
WEST CANADA LAKE—NO. 49 [2]					
Lake trout	6R569	M	53.3	1.268	0.91
	6R568	F	38.6	.426	1.14

NOTE: — indicates data unknown.
[1] Specific location within waters, if known, given in parentheses.
[2] Numbers refer to location of water in Fig. 1.
[3] Imm. = immature.
[4] One fillet.

TABLE 4.—*Total selenium in Cayuga Lake trout by age of trout, 1970*

Age (Years)	Selenium (ppm)
1	0.53
2	0.52
3	0.40
4	0.29
5	0.40
6	0.52
7	0.49
8	0.61
9	0.44
12	0.52

NOTE: Sensitivity level = 0.1 ppm.

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Effects of Insecticides on Populations of Rodents in Kansas—1965-69 [1]

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ABSTRACT

The general objective of this study was to determine the effect of recommended applications of commonly used insecticides on population dynamics of unconfined rodents in two sites in Ellis County, Kans. Six insecticides (diazinon, endrin, heptachlor, parathion, methyl parathion, and aldrin) were applied to one field, while a second field served as a control. Insecticide applications began in summer 1965 and continued through summer 1968. Live trapping of rodents was conducted on the treated field and the control field from 1965 through 1969. Specimens were collected each month for residue analysis.

During 19 separate 10-day trapping periods, 4,661 rodents were captured of which 162 were analyzed for the six insecticides. Insecticidal residues were detected in 36 of these specimens. Dieldrin residues were detected in 33 specimens and heptachlor epoxide in 8 specimens (five of these specimens had residues of both dieldrin and heptachlor epoxide). These residue levels were low with no dieldrin levels in specimens exceeding 0.50 ppm, and no heptachlor epoxide levels exceeding 0.02 ppm. No residues of the other four insecticides were detected in any of the specimens.

Species composition of the trapped rodents was similar for the two study areas as were the population levels, with Peromyscus maniculatus *comprising about 74% of the rodent population on the two areas. Population fluctuation trends for the two areas were also similar. Average minimal longevity for* P. maniculatus *was 45.7 days on the treated area and 50.9 days on the control area. Monthly survival between June and September was about 45% on both areas. Recaptures of* P. maniculatus *the year following their initial marking were slightly greater on the untreated area than on the treated area. The slight differences in dynamics of the rodent populations on the two areas could easily have been due to chance, minor differences between the two study areas, or differential emigration of specimens from the areas as well as direct or indirect effects of insecticidal applications.*

Introduction

Although much research has been done on various aspects (persistence, toxicity, species susceptibility, biological accumulation, etc.) of insecticides, little effort has been expended to study the long-term effects of insecticides on populations of unconfined animals. This lack of research is even more apparent when one tries to ascertain the effects of recommended insecticidal applications on wildlife populations under natural conditions. Because of this lack of fundamental data, research on the effects of recommended insecticidal applications on the population dynamics of unconfined small rodents was initiated in west central Kansas in 1965, as a subunit of a larger research project involving studies of fish populations, and insecticidal residues in soil, water, and vegetation. The results of the rodent population dynamics investigation are reported in this paper, while the results of other aspects of the comprehensive study were published separately in an earlier issue of the *Pesticides Monitoring Journal* (*14*).

The Study Area

This study was conducted on two sites in Ellis County, approximately 21 km (13 miles) southwest of Hays, Kans. Because the region was part of a newly created Cedar Bluffs Irrigation District and had not been intensively cultivated prior to 1965, insecticide use on the locale had been minimal. No insecticidal residues were detected in samples of water, soil, plants, or animals collected from the two study sites before the study began in 1965. The history and development of the general region are described in detail by Knutson *et al.* (*14*).

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Average annual rainfall for the area is 59 cm (23 in), of which 75% normally occurs in localized convective storms during the growing season (April through September). The mean annual temperature is 12°C (54°F), with mean monthly extremes of 27°C (80°F) for July and −2°C (29°F) for January. Silty clay loam is the most common soil type on the study sites. A more detailed discussion of climate and physical characteristics of the study area is presented in Leonard (*16*) and Knutson *et al.* (*14*).

The two sites utilized during this study were a "treated area" which had insecticides applied to it at recommended levels and an "untreated area" which had no insecticides applied to it. The treated area was a 7.0-hectare (ha) (19.5 acre) field with three terraces. The untreated area consisted of a similar 8.4-ha (22.7 acre) field with one terrace and situated 1.6 km (1 mile) south of the treated area.

Corn (*Zea mays*) and sorghum (*Sorghum vulgare*) were produced on both areas during the study period. The density and composition of vegetation on terraces and areas bordering both fields were similar and consisted mainly of small kochia (*Kochia scoparia*), dandelion (*Taraxacum officinale*), giant foxtail (*Setaria faberii*), yellow foxtail (*Setaria lutescens*), goldenrod (*Solidago* spp.), ragweed (*Ambrosia* spp.), and sandbur (*Cenchrus pauciflorus*).

Materials and Methods

SAMPLING PROCEDURES

Insecticidal applications on agricultural crops grown on the treated area were carefully monitored and were in accordance with recommended dosages and procedures of the Extension Entomologists of the Kansas Extension Service. Application rates varied from year to year and reflected the general insecticidal usage pattern of the region. A summary of the amounts of diazinon, endrin, heptachlor, parathion, methyl parathion, and aldrin applied to the treated area is presented in Table 1. The dates of insecticidal applications appear in Table 2. A more detailed discussion of time and method of insecticidal application on the treated area appears in Knutson *et al.* (*14*). No insecticides were intentionally applied to the untreated area during the course of the study, however, some dieldrin-treated sorghum seeds may have been planted inadvertently on the untreated area during one or two of the years.

TABLE 1.—*Summary of insecticides applied to the treated area (pounds per acre) from 1965 to 1968*

Year	Diazinon	Endrin	Heptachlor	Parathion	Methyl parathion	Aldrin
1965	0.60	0.00	0.00	0.38	0.00	2.45
1966	2.29	0.30	0.67	1.40	0.38	1.20
1967	2.13	0.36	0.43	0.79	0.50	0.84
1968	3.46	0.28	0.41	1.01	0.53	0.30

TABLE 2.—*Dates of insecticide application to the treated area*

Treatment [1]	Dates of Application			
	1965	1966	1967	1968
Soil	May 10-11	May 12	May 15-16	May 25
Foliage	Aug. 19	Aug. 3	Aug. 10	Aug. 5

[1] Diazinon, heptachlor, parathion, and aldrin were used as soil treatments while foliage was treated with diazinon, endrin, and methyl parathion.

Data on small rodent populations on the two areas were obtained by the capture-recapture method. Live traps similar to those of Scheffer (*19*) were placed at 18-m (60-ft) intervals on the terraces and along the edges of each study area; 215 live traps were used throughout the study, 151 on the treated area and 64 on the untreated area. More traps were set on the treated area than on the untreated area because of extra terraces in the treated area.

Initial trapping in August 1964 determined that rodents lived mainly in habitats along field edges and on the terraces of the two areas and moved laterally out into the fields foraging for food. At that time, 50 traps located within 60 feet of a terrace captured 21 rodents (42% success), while 50 traps located 120 feet or more from a terrace captured only 3 rodents (6% success). The soil under the crops was bare and weedless, affording little cover in which rodents could reside. Thus, trapping intensity, i.e., traps per 305 m (1,000 ft) of rodent habitat, was similar for both areas during the course of this study. Trapping was conducted on each area during 19 separate 10-day trapping periods, spring and summer 1965-1969. To reduce heat induced mortality of trapped animals, which could occur during daytime, all traps were baited and set at dusk, then emptied and/or sprung the following morning, and left unset until dusk.

Traps were baited with a mixture of rolled oats and peanut butter (*4*). Bait from the previous day was removed before a new ball of the mixture, approximately 13 cm (0.5 in) in diameter, was placed in each trap. Additional bait was used during cold weather (*11*).

Trapped animals were identified, marked, and released at the capture site. Toe clipping and ear notching combinations (*2*) were used to mark captured animals. To facilitate analysis, all data were placed on computer input cards following a format similar to Brotzman and Giles (*3*). All data were analyzed on an IBM 360/50 computer.

Population estimates were made using capture-recapture data and the procedures of Schnabel (*20*) and Schumacher and Eschemeyer (*21*). Compensation was made for differences in available trapping areas on the two study sites by calculating population estimates for each 305-m (1,000 ft) unit of trapline. There were 8.7 such

units on the treated area and 3.8 units on the untreated area. An index to annual mortality was determined by dividing the number of June and July recaptures of animals marked during the previous year by the total number of animals marked in the previous year.

During the last night of each trapping period of 1965 to 1968, five rodent specimens from each study area were collected for insecticidal residue analysis.

ANALYTICAL PROCEDURES

The specimens for insecticide analysis were quick frozen the morning they were captured. Prior to analysis for insecticidal residues, each specimen was allowed to thaw before being homogenized in a Warning high-peed blender. After homogenization, a 10-g sample of each specimen was analyzed separately for insecticidal residues. The 10-g sample was placed in an Omnimixer with 50 ml of redistilled hexane and sufficient anhydrous sodium sulfate to take up the water. The mixture was then blended at high speed for 1-2 minutes, then decanted through No. 43 Whatman filter paper into a 100-ml suction flask. The residue was extracted with two additional portions of hexane as described above, then filtered and combined in the suction flask. The container, filter paper, and contents were then washed with a final portion (10 ml) of hexane. The total hexane extract was transferred to a round-bottomed flask for concentration under vacuum at 35°-40°C to 2-3 ml of hexane, then transferred quantitatively to a 15 ml-centrifuge tube using 5 ml of hexane. An aliquot was used for cleanup and gas chromatographic analysis. All solvents were glass-distilled and purified for gas chromatographic analysis.

Cleanup methods for extracts, recoveries of insecticides from fortified samples, and sensitivities by gas chromatographic analysis are reported by Kadoum (*12,13*). Insecticide residues were not corrected for recovery with the exception of methyl parathion, since recovery from fortified samples was essentially 100%. The insecticides were separated and detected by electron capture gas chromatography using a 6-ft glass column, packed with 3% DC-11 on 60/80 mesh silanized Gas Chrom P (Applied Science Labs, State College, Pa.). Operating conditions were as follows: carrier gas—nitrogen at a flow rate of 36 ml/min; column temperature—200°C; injection temperature—240°C; detector cell—220°C; volume of extract injected—4 μl. The analytical procedure was capable of detecting residues of as low as 0.01 ppm of diazinon, parathion, methyl parathion, malathion, endrin, aldrin, dieldrin, heptachlor, heptachlor epoxide, *p,p'*-DDE, *o,p'*-DDT, and *p,p'*-DDT.

Results

Nineteen 10-day trapping periods on the two study areas produced data from 40,850 trap-nights (number of trapping periods × number of nights per period × number of traps set), 28,690 on the treated area and 12,160 on the untreated area. Totals of 3,306 and 1,355 individual rodents were captured on the treated and untreated areas, respectively. Fifty-four percent were recaptured later, and 35.8% of the individuals were recaptured more than once. Data from 17 trapping periods during the summers of 1965-68 were used to estimate rodent populations on the two areas. June and July recaptures of mammals marked during the previous summer were used to estimate annual mortality and survival for 1965-68. Two additional 10-day trapping periods were conducted in June and July of 1969 to estimate annual mortality and survival for the 1968-69 period.

P. maniculatus was the predominant species on both areas (Table 3). During the entire study, *P. maniculatus* made up 74.2% of the captures on the treated area and 74.0% of those on the untreated area. *Mus musculus* was the next most abundant small mammal trapped, contributing 7.4% and 8.6% to the total captures on the treated and untreated areas, respectively. *Sigmodon hispidus* was the third most commonly trapped mammal constituting 7.5% and 5.1% of the total captures on the treated and untreated areas, respectively. *Onychomys leucogaster*, *Microtus ochrogaster*, and *Reithrodontomys megalotis* each contributed less than 5.0% to the number of mammals trapped on either of the two areas. *Reithrodontomys montanus*, *Perognathus flavus*, *P. flavesens*, *P. hispidus*, *Spermophilus tridecemlineatus*,

TABLE 3.—*Species composition of 3,306 and 1,355 mammals captured on the treated and untreated area, respectively, during this study*

Species	Species Composition of Captures—(Percent)											
	1965		1966		1967		1968		1969		1965-69	
	TA	UA	TA	UA	TA	UA	TA	UA	TA	UA	TA	UA
Peromyscus maniculatus	64.1	68.4	64.8	63.6	83.0	85.5	78.9	73.7	82.3	81.5	74.2	74.0
Mus musculus	16.6	14.8	9.1	10.4	3.0	7.2	4.3	6.1	3.3	0.4	7.4	8.6
Onyshomys leucogaster	3.3	0.0	1.0	8.2	0.7	1.2	5.8	5.3	4.0	13.5	2.7	4.6
Sigmodon hispidus	3.5	4.8	11.4	10.8	9.0	0.9	7.2	6.6	3.1	0.4	7.5	5.1
Microtus ochrogaster	4.7	8.9	10.8	3.3	0.0	0.0	0.9	1.8	2.0	0.7	3.7	3.2
Reithrodontomys megalotis	1.5	2.1	0.5	1.3	3.8	1.5	0.9	0.9	4.7	2.2	2.1	1.5
Other species [1]	6.1	0.9	2.4	2.3	0.3	3.5	1.9	5.6	0.7	1.4	2.3	2.9

NOTE: TA = Treated Area; UA = Untreated Area.

[1] Other species included *Reithrodontomys montanus*, *Perognathus flavus*, *P. flavesens*, *P. hispidus*, *Spermophilus tridecemlineatus*, and *Dipodomys ordii*.

and *Dipodomys ordii*, each constituted less than 1.0% of the mammals trapped on either of the two areas.

Early in the analysis of data, attempts were made to estimate populations of each species of rodent on the area, but period-to-period and season-to-season variations were extremely large (Table 3). Therefore, population estimates of individual species were abandoned in favor of crude estimates of the composite rodent population and an estimate of the most abundant rodent species, *P. maniculatus*. The trappable populations estimated by the Schumacher-Eschemeyer method were slightly higher than those calculated by the Schnabel method. The differences in estimates by these two methods averaged 3.4% (range of 0.0% to 18.4%) for the entire study, and this difference was not significant for the entire study nor for any specific trapping period during the study.

Insecticidal residues were detected in 36 of 162 specimens analyzed. Residues of dieldrin were detected in 33 specimens and heptachlor epoxide in 8 specimens (5 specimens contained residues of both dieldrin and heptachlor epoxide). No residues of the other four insecticides were detected in any of the specimens. Residue concentrations were low ranging from 0.01 to 0.50 ppm of dieldrin and 0.01 to 0.02 ppm of heptachlor epoxide. Seven (7.1%) of 98 specimens from the treated area had detectable residues of heptachlor epoxide, while only 1 (1.6%) of 64 specimens from the untreated area showed heptachlor epoxide residues. Residues of dieldrin were detected in 26 (27.6%) of the 98 specimens from the treated area and in 7 (10.9%) of the 64 specimens analyzed from the untreated area (Table 4). Average dieldrin residue concentrations (0.07 ppm) were higher in specimens from the treated area than the average residue concentrations (0.04 ppm) in specimens from the untreated area.

Of nine species of rodents analyzed for insecticidal residues, five contained detectable amounts of dieldrin (Table 5). Specimens of *Mus musculus*, *Spermophilus tridecemlineatus*, and *P. maniculatus* contained more dieldrin (average concentrations of 0.07 to 0.10 ppm) in their carcasses than the other specimens. *Onychomys leucogaster* and *Reithrodontomys megalotis* also contained dieldrin residues in their carcasses, but in lesser amounts (average concentrations of 0.03 ppm). With the exception of specimens of *Reithrodontomys megalotis*, average residue concentrations were greater in carcasses of species collected from the treated area than those from the untreated area. Heptachlor epoxide residues (0.01 to 0.02 ppm) were found in four specimens of *P. maniculatus* and in one each of *Onychomys leucogaster*, *Sigmodon hispidus*, and *Spermophilus tridecemlineatus* from the treated area. One *Onychomys leucogaster* from the untreated area had residues (0.01 ppm) of heptachlor epoxide. Of the 36 specimens in which residues were detected, 20 (56.0%) were males and 16 (44%) were females.

TABLE 4.—*Summary of dieldrin residues (≥0.01 ppm) in 162 collected during this study*

Year	Treated Area						Untreated Area					
	No. Analyzed	Dieldrin Residues Present					No. Analyzed	Dieldrin Residues Present				
		No. Positive	Percent Positive	Concentrations (ppm)				No. Positive	Percent Positive	Concentrations (ppm)		
				Average	Median	Range				Average	Median	Range
1965	19	3	15.8	0.09	0.01	0.01-0.24	14	0	0.0	—	—	—
1966	22	3	13.0	0.08	0.09	0.02-0.13	18	3	16.7	0.02	0.02	0.01-0.03
1967	31	14	45.2	0.05	0.02	0.01-0.50	12	2	16.7	0.01	0.01	0.01-0.01
1968	26	6	23.1	0.13	0.03	0.01-0.44	20	2	10.0	0.09	0.09	0.04-0.15
Total	98	26	27.6	0.07	0.02	0.01-0.50	64	7	10.9	0.04	0.02	0.01-0.15

TABLE 5. *Dieldrin residues (≥0.01 ppm) in nine rodent species analyzed during this study*

Species	Treated Area						Untreated Area					
	Number Analyzed	Number Contaminated	Percent Contaminated	Concentration in Those Contaminated (ppm)			Number Analyzed	Number Contaminated	Percent Contaminated	Concentration in Those Contaminated (ppm)		
				Average	Median	Range				Average	Median	Range
Peromyscus maniculatus	31	12	38.7	0.08	0.01	0.01-0.50	21	3	14.3	0.03	0.03	0.01-0.04
Mus musculus	14	6	42.9	0.10	0.01	0.01-0.44	11	2	18.2	0.08	0.08	0.02-0.15
Onychomys leucogaster	9	3	33.3	0.03	0.02	0.02-0.05	7	1	14.3	0.01	—	—
Reithrodontomys megalotis	6	1	16.7	0.03	—	—	5	1	20.0	0.03	—	—
Spermophilus tridecemlineatus	7	4	57.1	0.07	0.02	0.01-0.24	0	0	0	—	—	—
Sigmodon hispidus	15	0	0	—	—	—	6	0	0	—	—	—
Microtus ochrogaster	9	0	0	—	—	—	8	0	0	—	—	—
Perognathus flavescens	5	0	0	—	—	—	2	0	0	—	—	—
Perognathus hispidus	2	0	0	—	—	—	4	0	0	—	—	—

Fluctuations of the total rodent populations on the treated and untreated areas followed the same general trend during the study (Fig. 1). The longest time between the date of first capture and date of last capture of a specific specimen was 34 months, a *Sigmodon hispidus* male on the treated area. Two *P. maniculatus* males survived at least 24 months on the treated area. Several individuals were known to have survived at least 14 months on the untreated area. For *P. maniculatus*, the average time between date of initial capture and date of last recapture varied from a high of 53.4 days on the untreated area to a low of 42.8 days on the treated area (Table 6). The average minimal longevity during 1965-68 for *P. maniculatus* on the treated area was less than the average minimal longevity for *P. maniculatus* on the untreated area.

FIGURE 1.—*Estimated (Schumacher-Eschemeyer method) trappable population per 305 m (1,000 ft) of trapline on the treated and untreated areas. Vertical lines represent 95% confidence intervals about the mean.*

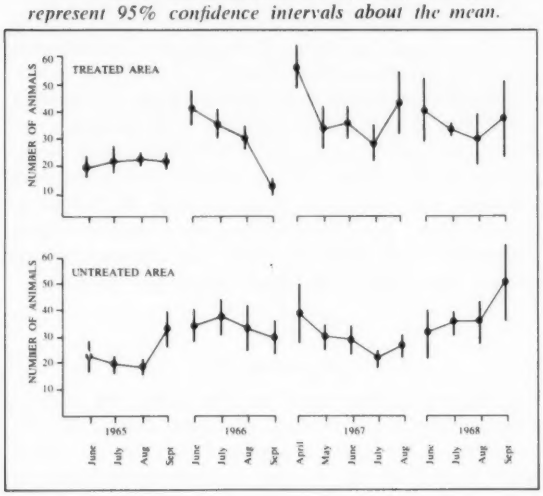

TABLE 6.—*Average minimal longevity (±S.D.) for* Peromyscus maniculatus *recaptured during this study*

Year [1]	Treated Area (days)	Untreated Area (days)
1965	48.1 ± 7.8	49.8 ± 8.7
1966	46.7 ± 11.4	50.6 ± 14.2
1967	42.8 ± 6.2	53.4 ± 8.9
1968	45.2 ± 5.8	49.9 ± 9.1
Mean (± S. E.)	45.7 ± 2.3	50.9 ± 1.7

[1] Indicates year in which the individuals were first captured.

An index to summer mortality within each rodent population was determined by following a single group of marked individuals through a summer. Rodents marked during the June trapping period of each year on each area were considered separate groups and later recaptures of these rodents were used as evidence of survival. Life tables were constructed to obtain an index to mortality (*10*). The index to monthly mortality was strikingly similar for both areas, averaging 54.5% and 54.9% on the treated and untreated areas, respectively (Table 7).

The number of animals marked one year and recaptured during June or July the following year was small, averaging 3.4% and 5.9% for the treated and untreated areas, respectively (Table 8).

TABLE 7.—*Summer mortality of* Peromyscus maniculatus *on the two study areas determined from recaptures of* P. maniculatus *marked in the initial trap period of each year*

Year	Number Marked in June Period	Number Recaptured in Succeeding Months			
		July	August	September	Total
TREATED AREA					
1965	310	194	173	154	521
1966	389	204	176	155	535
1967	493	175	154	—	329
1968	376	188	164	142	494
Totals	1,568	761	667	451	1,879
Marked *P. maniculatus* Available		1,568	1,568	1,075	—
Number Recaptured		761	667	451	1,879
Number Alive at Start		1,879	1,118	451	3,448
Percent Monthly Mortality		40.5	59.7	100.0	54.5
UNTREATED AREA					
1965	107	60	54	47	161
1966	112	56	50	44	150
1967	128	57	50	—	107
1968	100	48	42	34	124
Totals	447	221	196	125	542
Marked *P. maniculatus* Available		447	447	319	—
Number Recaptured		221	196	125	542
Number Alive at Start		542	321	125	988
Percent Monthly Mortality		40.8	61.1	100.0	54.9

TABLE 8.—Peromyscus maniculatus *recaptured during June and July trapping periods the year following initial marking on the two areas*

Year Marked	Treated Area		Untreated Area	
	Number Marked	Recaptures Succeeding Year	Number Marked	Recaptures Succeeding Year
1965	319	10	160	5
1966	437	21	165	16
1967	733	18	274	14
1968	502	18	192	12
Total	1,991	67	791	47
Percent Recaptured		3.4		5.9

Discussion

This study was meant to be a comparison of two rodent populations; one exposed to insecticides and one not exposed to insecticides. However, the experimental design did not permit replications or a study of two closed populations. As far as could be discerned from field observations, the only fundamental difference between

the two study areas which could have markedly influenced rodent populations was the insecticide treatment. Since the populations were not closed, rodent immigration and emigration were unknowns in the study. Likewise since no large exclosures were used, natural predation on the two areas could not be controlled. It appears though that the rodent populations on the two areas were very similar in composition and dynamics. Without adequate replications, only major changes in the rodent populations could have been detected. Since no attempt was made in this study to accurately determine the total population of rodents on each of the two study areas, the many limitations of the basic capture-recapture method discussed by Manly (*18*), Cormack (*5*), Tanton (*22*), and Leslie (*17*) did not influence the results. The same methods of trapping and marking were used on both study areas, thus, although the population estimates could not be regarded as entirely unbiased, the estimates are comparable. The population parameters calculated from these capture-recapture data are likewise comparable.

The relative abundance of some mammals changed from year to year, but since these changes occurred on both areas, these changes could not be considered insecticide induced. Year-to-year changes in population density and species composition are not uncommon in prairie regions of Kansas (*6-9*).

The average longevities calculated from capture-recapture data in this study are minimal values since they represent the time between first and last capture of the animal, not between birth and death. Unless there are differences in emigration rates of rodents from the two areas, these minimal longevity figures are comparable. Longevity of *P. maniculatus* on the treated area was slightly less than that of the population on the untreated area. Because the difference was slight and egress from the two populations not controlled, it is doubtful that the 5.2-day difference in longevity can be specifically attributable to insecticide applications on the treated area.

An index to mortality of *P. maniculatus* during summer months as estimated from recaptures was almost identical on the two areas. Again, unless there was differential emigration and trap vulnerability on the two areas, these indices to mortality are comparable. Carryover of marked *P. maniculatus* from one trapping season to the next may have been a bit greater on the untreated area, but this could not be confirmed since emigration was not controlled.

The actual pathway of insecticidal contamination was not studied; however, the rodents probably acquired insecticides through their food. *Sigmodon, Microtus,* and *Perognathus* feed primarily on plant material and seeds and had no insecticide residues in their carcasses while rodents (*Peromyscus, Mus, Onychomys, Reithrodontomys,* and *Spermophilus*) which had a diet that included insects in addition to seeds and plant material had insecticide residues in their carcasses. Contaminated insects (dead, dying, or those containing low levels of insecticides) could have been a source of contamination for insect eating rodents in our study as was found for insect eating rodents in Missouri (*15*).

Probably the most significant findings of this study were the low incidence and levels of contamination observed in the rodent population on the treated area. Only one species, *Spermophilus tridecemlineatus,* exhibited an incidence of contamination exceeding 50%. *P. maniculatus,* the most abundant species on the area, had an incidence of contamination of only 38.7% on the treated area. Levels of contamination never exceeded 0.50 ppm of dieldrin for any specimen analyzed. Although levels of dieldrin contamination were higher in rodents from the treated area early in the study, levels of contamination increased on the untreated area during the last year of our study. Collection of contaminated specimens on the untreated area was a result of immigration onto that area by rodents from surrounding areas; similar movements probably were occurring on the treated area. It appears therefore that unless closed populations of rodents can be assured, their population dynamics are probably a poor indicator of insecticide contamination. It is doubtful that monitoring of rodent populations can detect anything but changes of extremely large magnitude under unconfined conditions.

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See Appendix for chemical names of compounds discussed in this paper.

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Mercury Residues in Fish From Saskatchewan Waters With and Without Known Sources of Pollution—1970 [1]

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ABSTRACT

In 1970, mercury concentrations were determined in muscle tissue of 125 fish from 10 Saskatchewan waters, 6 of which were part of the Saskatchewan River system receiving industrial and/or municipal wastes containing mercury and 4 of which had no known sources of mercury pollution.

Concentrations in fish from the Saskatchewan River system ranged from 0.18 to 8.88 ppm depending on the degree of pollution of the fish collection site. Levels were higher (0.18 to 8.88 ppm) in fish caught downstream from a chlor-alkali plant in Saskatoon than those (0.25 to 1.2 ppm) in fish from sites upstream from this plant whose sources of mercury were municipal or industrial wastes other than the chlor-alkali industry. Levels in fish from the four lakes without any known source of mercury pollution ranged from 0.11 to 1.13 ppm. These results indicate that mercury levels in fish from apparently unpolluted waters may exceed the Canadian actionable level of 0.5 ppm for mercury; such residues may be of natural origin.

Introduction

Both nature and man contribute to the contamination of our environment with mercury (*7*). It has been estimated that the surface of the earth receives about 100,000 tons of mercury annually from precipitation (*14*) as compared to the world production of 10,000 tons of mercury per year (*16*). The earth's crust contains an average of 0.5 ppm mercury (*6*), and traces of natural mercury are to be expected everywhere; for example, mercury was found in sea water as early as 1777 (*11*). Subsequent studies by Stock and Cucuel (*15*) during the 1930's showed that mercury was truly ubiquitous, being present in air, water, foods, and soils. Man by his activities has increased the concentration of mercury in some foods, particularly in fish, in some localized areas. The consequence of the pollution of water with mercury-containing industrial effluent was evident in Minamata, Japan, where in the 1950's several persons died or became seriously ill after eating fish or shellfish containing very high levels (27-102 ppm, dry-weight basis) of mercury (*10*).

To understand the contamination of fish with mercury from human activities, the various uses of mercury have to be considered. Of the 3,006 tons of mercury used in the United States in 1969, 1,156 tons (38.4%) were used in the chlor-alkali industry, 710 tons (23.6%) in the electrical industry, 116 tons (3.8%) in dentistry, 102 tons (3.4%) in agriculture, 370 tons (12.3%) in the paint industry, and the rest (18.5%) in laboratories, pulp and paper, pharmaceutical, and other industries (*3*). In the same year, 150 tons were used in Canada: 100 tons (66.7%) in the chlor-alkali industry and 12.5 tons (8.3%) in agriculture (*8*). Some mercury used in chlor-alkali and other industries escapes into local waters. Large urban centers may also contribute to water pollution with mercury since a significant amount of mercury is used in paints, hospitals (e.g., thermometers), laboratories, dental preparations, electrical equipment, batteries, light-bulbs, etc., some of which may enter sewage and become part of local waters. Elevated mercury levels are to be expected in fish from waters that receive mercury-containing industrial and/or municipal wastes.

The use of mercury in agriculture, mainly as seed-dressings, accounts for only a small part of total mercury consumption. Furthermore, mercurial seed-dressings are

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used at about 500 mgHg/acre or 0.00055 ppm/6-inch acre (*13*). Such small quantities of mercury in a vast land area should not appreciably contribute to the pollution of water, as this amount, even after 30 years of continuous use on the same land, adds up to only 0.016 ppm. This amount is insignificant when compared to the natural mercury content of soil (*2,15*) and is less than the amount of mercury received by soil through precipitation (*2*). Thus, it is perhaps safe to assume that the mercury in fish from waters in agricultural areas not receiving any industrial and/or municipal wastes can be considered to be of natural origin.

Numerous studies from many countries have reported the mercury content of fish from contaminated waters (*12*). Mercury levels in fish from many rivers and lakes in Canada and the United States have been reported in excess of 1 ppm (*1,4,5,17*), and commercial fishing in many waters has been banned. Little, however, is known about the mercury content of fish from apparently unpolluted waters. Studies of "background" levels would be useful in establishing a practical actionable level since such levels of natural mercury have probably been present in fish all the time and may have done man no harm.

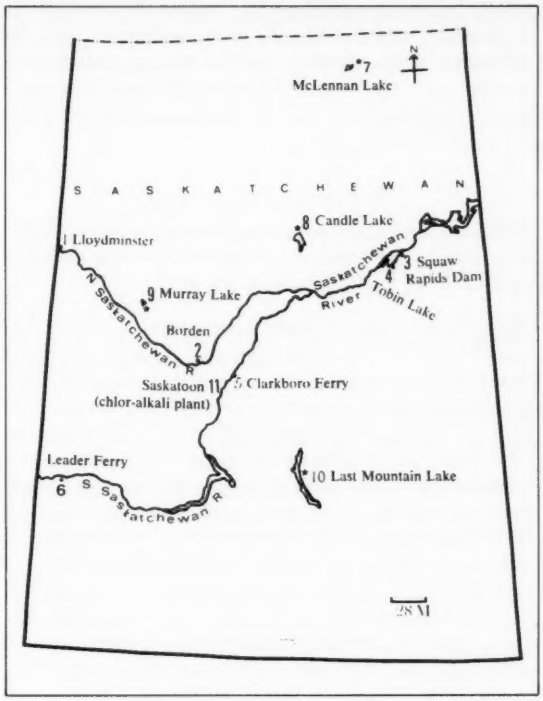
The object of the study reported here was to determine the mercury levels in fish from 10 Saskatchewan waters, 6 with industrial and/or municipal sources of mercury pollution (Group I sites) and 4 with essentially only natural sources of mercury (Group II sites).

Sampling Sites and Procedures

The fish collection sites are shown in Fig. 1. Group I sites were located on the Saskatchewan River system and included Lloydminster (Site 1) on the North Saskatchewan River at the Alberta-Saskatchewan border; Borden (Site 2); Squaw Rapids Dam (Site 3); Tobin Lake (Site 4); Clarkboro Ferry (Site 5); and Leader Ferry (Site 6). The city of Saskatoon (designated Site 11) is a principal source of pollution, since a chlor-alkali plant there discharges effluents containing mercury into the river system; all sampling sites downstream from Saskatoon would be contaminated by this plant as well as by municipal wastes from Saskatoon and several other cities upstream. Sampling sites upstream from Saskatoon and on the South Saskatchewan River would be contaminated only by cities at appreciable distance upstream, such as Calgary in Alberta; Edmonton, also in Alberta, may contribute to pollution of the North Saskatchewan River.

The Group II sampling sites, McLennan Lake (Site 7), Candle Lake (Site 8), Murray Lake (Site 9), and Last Mountain Lake (Site 10), did not receive industrial or municipal wastes. McLennan Lake in Northern Saskatchewan is remote from human activities, while

FIGURE 1.—*Map of Saskatchewan showing the collection sites—1970*



agricultural land forms a substantial part of the precipitation area of Candle Lake, and Murray and Last Mountain Lakes are within agricultural areas. It has been pointed out that agricultural uses of mercury do not contribute to pollution of water or fish (*12*). Although water or bottom mud from the lakes were not analyzed, these Group II sites were considered apparently unpolluted by man.

During the summer of 1970, a total of 125 fish were netted at the 10 sampling locations and brought to the laboratory as rapidly as possible. Muscle tissue samples were taken from the longitudinal dorsal muscles on the anterior section of each fish and frozen in plastic bags at −18° C until analyzed.

Analytical Procedures

The total mercury content of each muscle tissue sample was determined by atomic absorption spectrometry according to the method of Saha *et al.* (*13*). A 4- to 5-g sample of fish tissue was digested under reflux with concentrated nitric acid and perchloric acid to destroy organic matter. The acidity of the digest was adjusted to approximately 1N, and hydroxylamine hydrochloride was added to destroy excess oxidants. The solution was then extracted twice with chloroform to remove any

organic matter. Mercuric ions were then extracted with a chloroform solution of dithizone and determined by atomic absorption spectrometry. About 93 to 98% of the mercury added to muscle tissue as $HgCl_2$ could be recovered by this method, and the minimum detectable amount was 0.005 ppm Hg. All fish specimens were analyzed in duplicate, and the mercury levels were not corrected for recovery.

Results and Discussion

The average mercury concentration with standard deviation, range, and median values for fish species from each sampling site are given in Table 1.

The mercury concentrations in fish from Group 1 sites ranged from 0.18 to 8.88 ppm and generally averaged above the 0.5 ppm Canadian actionable level. The goldeye fish from Lloydminster (Site 1) averaged 0.74 ppm of mercury; and long-nose suckers from Borden (Site 2), a point downstream from Lloydminster, had a rather high average level for the species (0.57 ppm) since this species is known to accumulate less mercury than other species, for example, pike. The waters at those two sites on the North Saskatchewan River were probably contaminated by industrial and municipal wastes from cities along the river bank. Squaw Rapids Dam (Site 3) and Tobin Lake (Site 4) were only 30 miles apart and both received contaminated waste from the chlor-alkali plant in Saskatoon (about 300 miles upstream) and other industrial and municipal wastes from many cities along the river system. The average mercury content of pike varied considerably for these two sites (Site 3—1.94 ppm and Site 4—0.86 ppm) although the lower ranges were similar (Site 3—0.49 ppm and Site 4—0.63 ppm). These results are not unusual, since other studies have reported widely differing amounts of mercury in the same species from the same location (*9,17*). The highest average mercury concentrations were in goldeye (2.05 ppm) and pike (4.80 ppm) from Clarkboro Ferry (Site 5); these levels reflect the effects of direct discharge of individual wastes containing mercury into water, since the sampling site is only a few miles downstream from the chlor-alkali plant in Saskatoon (Site 11).

It has been estimated that about 0.25 to 0.5 lb mercury can be lost to the environment for every ton of chlorine produced, and plants producing 100 tons or more of chlorine per day are rather common (*3,4*). A plant of this size can then discharge 9 to 18 thousand lb of mercury per annum into the environment, mainly in water.

Fish (Sauger) from Leader Ferry (Site 6), a site upstream from the chlor-alkali plant in Saskatoon and contaminated with municipal or other types of industrial wastes averaged only 0.67 ppm mercury.

The mercury concentrations in fish from Group II sites ranged from 0.11 to 1.13 ppm. Mean levels of mercury were low in walleye from Candle Lake and pike from McLennan Lake, averaging 0.18 ppm and 0.24 ppm of mercury, respectively. The walleye from Last Mountain Lake, however, averaged 0.45 ppm, a concentration barely within the 0.5 ppm Canadian actionable level. Of particular interest was the mercury content of perch from Murray Lake (0.71 ppm), a level about three times higher than in perch from Tobin Lake (0.24 ppm), a site with known industrial and municipal mercurial contamination. These results indicate that mercury levels in fish in excess of 0.5 ppm do not necessarily result

TABLE 1.—*Mercury residues in fish from some Saskatchewan waters with and without known sources of pollution—1970*

Sampling Site No. and Location	Fish Samples	No. of Fish	Mercury Concentration (PPM, Fresh-Weight Basis)		
			Mean ± S.D.	Range	Median
GROUP I (INDUSTRIAL AND/OR MUNICIPAL POLLUTION)					
1. Lloydminster [1]	Goldeye	10	0.74 ± 0.25	0.52-1.2	0.63
2. Borden [1]	Longnose sucker	15	0.57 ± 0.19	0.25-0.90	0.57
3. Squaw Rapids Dam [2]	Pike	10	1.94 ± 2.50	0.49-8.88	1.06
4. Tobin Lake [2]	Pike	10	0.86 ± 0.24	0.63-1.2	0.91
	Sauger	6	0.98 ± 0.26	0.79-1.50	0.89
	Goldeye	16	0.42 ± 0.08	0.30-0.55	0.41
	Perch	12	0.24 ± 0.07	0.18-0.42	0.21
5. Clarkboro Ferry [3]	Goldeye	5	2.05 ± 1.37	0.96-4.25	1.30
	Pike	3	4.80 ± 2.25	2.60-6.11	6.09
6. Leader Ferry [1]	Sauger	4	0.67 ± 0.20	0.42-0.84	0.71
GROUP II (NO KNOWN SOURCES OF POLLUTION)					
7. McLennan Lake	Pike	4	0.24 ± 0.07	0.16-0.32	0.23
8. Candle Lake	Walleye	9	0.18 ± 0.04	0.11-0.24	0.19
9. Murray Lake	Perch	11	0.71 ± 0.20	0.42-1.13	0.70
10. Last Mountain Lake	Walleye	10	0.45 ± 0.13	0.23-0.68	0.41
Total		125			

[1] Polluted with industrial and municipal wastes from cities, but no chlor-alkali plant.
[2] Polluted with industrial and municipal wastes and by effluents from a chlor-alkali plant at considerable distance upstream.
[3] Polluted with industrial and municipal wastes and direct discharge of effluents from a chlor-alkali plant.

from man's use of mercury, since the only sources of mercury for Group II sites are agricultural or natural levels in bedrock. High levels of mercury in fish from other apparently uncontaminated water have been reported earlier; Wobeser *et al.* (*17*) reported 0.8 ppm Hg in a lake trout from a Northern Canadian lake, and Johnels *et al.* (*9*) reported 0.75 to 1.1 ppm Hg in pike from several apparently uncontaminated waters in Sweden. Mercury content of bedrock was suggested as a possible source for such high levels (*9*). The bedrock in Saskatchewan is primarily sedimentary which, in general, has a higher mercury content than igneous rocks (*6*).

The results obtained in this study show that high levels of mercury (greater than 1 ppm) can be expected in fish from waters contaminated directly by industrial effluents containing mercury such as those from the chlor-alkali industry. The mercury content of fish can exceed 0.5 ppm if the source waters are contaminated with industrial and municipal wastes from urban centers even though they may have no chlor-alkali plant contamination. Lastly, the mercury content of fish from some apparently uncontaminated waters may exceed the 0.5 ppm Canadian actionable level; such high levels are probably of natural origin and related to the mercury content of the bedrocks.

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PESTICIDES IN SOIL

Pesticide Residues in Soil From Eight Cities—1969

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ABSTRACT

Soil samples from eight cities were analyzed for pesticide residues. Besides DDT and its metabolites (DDTR), other pesticides detected were dieldrin, chlordane, heptachlor, heptachlor epoxide, toxaphene, and endrin. No organophosphate residues were detected. Levels of DDTR varied significantly among the eight cities, with the highest average residue level in Miami, Fla. (5.98 ppm) and the lowest in Houston, Tex. (0.35 ppm). When residue levels in lawn or garden areas were compared to those in unkept areas within the cities, DDTR residues were significantly greater for lawn areas.

Introduction

Much information has been developed on pesticide residues in soils; however, little is available on residue levels in soil from urban areas. Fahey, Butcher, and Murphy (*1*) sampled urban soil in Battle Creek, Mich., for chlorinated hydrocarbons and found residues of DDT ranging from 0.07 to 79.92 ppm. Purves (*2*) studied the difference between residues of trace elements found in urban garden plots and those in rural areas and found residues of boron, lead, and zinc to be greater in urban garden plots than rural plots.

The present report covers a preliminary survey of pesticide residues in soil initiated in response to the need for more intensive studies of pesticide residues in urban areas of the United States.

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Sampling Procedures

Eight U.S. cities at different geographical locations throughout the country were selected for soil sampling during the summer and fall of 1969. Fifty sampling sites were randomly chosen within each city. Each site was a 50- by 50-ft plot, modified at times to meet local conditions, but always containing 2,500 square feet. Within each site, 9 soil cores (2 inches in diameter by 3 inches deep) were collected on a 3- by 3-grid. These cores were sifted through a ¼-inch mesh, composited, and shipped to the Pesticide Monitoring Laboratory in Gulfport, Miss.

Analytical Procedures

A subsample of soil, weighing 300 g wet weight, was placed in a 2-qt jar with 600 ml of 3:1 hexane-isopropanol solvent. The jars were sealed and rotated for 4 hours. After rotation, the soil was allowed to settle, and 200 ml of the extract solution was filtered into a 500-ml separatory funnel. The isopropanol was removed by two washings with distilled water. The hexane was then filtered through a funnel containing glass wool and anhydrous sodium sulfate (Na_2SO_4). No further cleanup was normally required before analysis.

Gas chromatography was employed for the analysis of organochlorine pesticides using the electron affinity detector and for organophosphorus pesticides using the flame photometric detector. The essential experimental conditions were as follows:

Gas Chromatographs:

1. Hewlett-Packard Model 402 and Model 810—tritium foil electron affinity detector (pulsed-cell potential); carrier gas—5% methane in argon and flow rate—about 80 ml/min.
2. MicroTek Model 220—tritium foil electron affinity detector (DC-cell potential); flame photometric detector, phosphorus filter, hydrogen-oxygen flame (hydrogen flow rate—150-200 ml/min, oxygen flow rate—30-35 ml/min); carrier gas—purified nitrogen and flow rate—about 100 ml/min

Columns:	Glass, 183 cm long by 6 mm, o.d. and 4 mm, i.d. with the following packings:	
	3% or 10% DC-200 on 100/120 mesh Gas Chrom Q	
	9% QF-1 on 100/120 mesh Gas Chrom Q	
	5% XE-60 on 100/120 mesh Chromosorb W	
Temperatures:	Detector	200° C
	Injection port	250° C
	Column DC-200	180° C
	Columns QF-1 and XE-60	160° C

Peak height was used to measure the amount of pesticide residues. A dual-column system employing polar and nonpolar columns was utilized to identify and confirm pesticides. Further confirmation, when necessary, was made by thin layer chromatography or partition values. Average recovery values for most of the organochlorine pesticides ranged from 90% to 100%, and with the exception of chlordane and toxaphene, the detection limits were 0.01 to 0.02 ppm. Chlordane was calculated using gamma-chlordane as a standard (gamma-chlordane constitutes an average of 10% of technical chlordane), with a detection limit of 0.04 ppm. Toxaphene was measured by comparing the summation of peak heights of several peaks (usually four selected peaks from its multiple-peak gas chromatogram) with the corresponding peaks of a known standard, and the detection limit was 0.05 ppm. All residues reported were corrected for recovery.

The present analytical methodology should detect common organophosphate pesticides such as DEF®, diazinon, EPN, ethion, azinphosmethyl, malathion, parathion, phorate, and carbophenothion. The recovery values ranged from 80% to 100%, with detection limits ranging from 0.01 ppm for phorate to 0.1 ppm for azinphosmethyl and EPN. It should be noted, however, that the present extraction technique would not recover many of the possibly oxidized metabolic products.

Arsenic was determined by atomic absorption spectrophotometry. The soil sample was first extracted with 9.6N hydrochloric acid and then reduced to the As^{+3} state with stannous chloride. As^{+3} was partitioned from the acid to benzene, then further partitioned from benzene into water for the absorption measurement. A Perkin-Elmer Model 303 instrument was used, and absorbence was measured with an arsenic cathode lamp at 1970 Å with argon as an aspirant to an air-hydrogen flame. This method was rapid and precise, but indicated an average recovery rate of 56% for soil samples. The arsenic levels reported in this study were corrected for the recovery value; the detection limit after correction, was 0.2 ppm.

Results and Discussion

Table 1 shows the average residues detected for all cities. Miami had the highest combined residues of DDTR (DDT and related degradation products). It also had the highest dieldrin and chlordane residues. The dieldrin residues were particularly outstanding, because the average residue in Miami was over 10 times greater than the next highest average dieldrin residue, which was detected in Bakersfield. The highest arsenic residues were found in Salt Lake City and the lowest, in Houston. No organophosphate residues were detected.

The range of residues and the percent of sites with detectable residues are also given in Table 1. Although Bakersfield, Calif., had an average residue of 0.36 ppm for DDTR, the range was narrow (0.02 - 3.08 ppm) and 94% of the sites had detectable residues. This indicates an equal distribution of DDTR residues over the entire city. In contrast, Waterbury, Conn., has an average DDTR residue of 0.98 ppm and a wide range (0.01 - 10.35 ppm), with only 56% of the sites having detectable residues, indicating an unequal distribution over the city.

With the exception of Houston and Manhattan, cities in this study were located in States where croplands had previously been sampled as part of the National Soils Monitoring Program. Residues of arsenic, DDTR, dieldrin, and chlordane, four commonly occurring pesticides, are compared in Table 2. In most cases, the average residues detected in the cities tended to be greater than the corresponding cropland residues; however, because no tests of significance can be made on the data at this time, caution should be exercised in interpreting the results in Table 2.

Pesticide residue data can usually be described by a log normal distribution. Frequently, however, the data contain a large number of zero values, resulting either from the absence of pesticides or their presence at levels below the analytical sensitivity. To include these zero values when taking logarithms, a value must be substituted for the zero figure. After repeated tests for significant kurtosis and/or skewness, the log (X+.01) transformation most closely approximated the normal distribution. Using this transformation, logarithmic means were determined for DDTR and arsenic residues. The 95% confidence interval about each of these means was determined. The antilogs of these figures were taken to give estimates of the geometric mean and its 95% confidence interval in the untransformed dimension. The results are given in Table 3. All figures are in parts per million.

For DDTR, the geometric mean for soil from Miami, Fla., (1.60 ppm) far exceeded means from the other cities; the confidence interval ranged from 0.91 to 2.82 ppm. Camden, N. J., had the second highest mean (0.37 ppm) with an upper limit of 0.62 ppm. Houston, Tex., had the lowest mean (0.02 ppm), and its upper limit (0.04 ppm) was the same as the lower limits for Manhattan, Kan., and Waterbury, Conn., which had virtually identical means and confidence intervals. The

TABLE 1.—*Pesticide residues in soil from eight cities, 1969*

City	Residues in PPM and Percent Positive Sites													
	Arsenic	*o,p'*-DDT	*p,p'*-DDT	*o,p'*-TDE	*p,p'*-TDE	*o,p'*-DDE	*p,p'*-DDE	DDTR	Dieldrin	Chlordane	Hepta-chlor	Hepta-chlor Epoxide	Toxaphene	Endrin
Bakersfield, Calif.														
Range	1.2-32.2	0.01-0.72	0.01-1.72	ND	0.01-0.33	0.01-0.16	0.01-0.92	0.02-3.08	0.01-1.98	0.07-20.48	0.02-0.10	0.05-0.18	ND	0.01-0.03
Average	7.1	0.05	0.15		0.03	0.01	0.12	0.36	0.07	0.78	<0.01	0.01		<0.01
Percent Positive sites [1]	100.0	66.0	90.0		70.0	32.0	94.0	94.0	28.0	30.0	4.0	8.0		6.0
Camden, N.J.														
Range	1.0-46.3	0.01-2.06	0.03-6.74	0.01-0.28	0.01-2.19	[2] 0.01	0.02-3.11	0.09-13.44	0.02-0.21	0.39-5.90	ND	0.02-0.39	ND	ND
Average	11.2	0.19	0.75	0.02	0.16	<0.01	0.23	1.36	<0.01	0.36		0.01		
Percent Positive sites [1]	100.0	66.0	88.0	34.0	78.0	2.0	84.0	88.0	4.0	16.0		6.0		
Houston, Tex.														
Range	0.2-15.3	0.01-3.06	0.01-3.79	[2] 0.07	0.01-0.49	0.01-0.06	0.01-0.90	0.01-7.68	0.01-1.47	0.04-12.94	0.01-0.02	0.01-0.14	[2] 0.11	[2] 0.01
Average	2.1	0.11	0.17	<0.01	0.02	<0.01	0.05	0.35	0.04	0.66	<0.01	<0.01	<0.01	<0.01
Percent Positive sites [1]	98.0	32.0	34.0	2.0	18.0	4.0	28.0	40.0	20.0	34.0	6.0	20.0	2.0	2.0
Manhattan, Kans.														
Range	0.7-72.0	0.01-1.24	0.01-4.65	[2] 0.01	0.01-0.78	0.01-0.03	0.01-1.53	0.01-8.20	0.01-0.72	0.03-4.86	0.02-0.09	0.01-0.44	[2] 12.07	ND
Average	11.5	0.11	0.45	<0.01	0.06	<0.01	0.15	0.78	0.04	0.30	<0.01	0.03	0.24	
Percent Positive sites [1]	100.0	48.0	56.0	2.0	50.0	4.0	54.0	58.0	20.0	40.0	10.0	26.0	2.0	
Miami, Fla.														
Range	0.3-33.9	0.01-5.91	0.04-42.56	0.01-0.27	0.01-5.06	0.01-0.15	0.01-11.15	0.01-52.38	0.01-8.58	0.04-16.87	0.01-0.02	0.01-0.45	14.79-52.73	ND
Average	2.3	0.79	2.67	0.02	0.41	0.01	2.09	5.98	0.72	1.59	<0.01	0.05	1.34	
Percent Positive sites [1]	80.0	86.0	94.0	24.0	84.0	8.0	100.0	100.0	64.0	64.0	6.0	50.0	4.0	
Milwaukee, Wis.[3]														
Range	1.2-54.4	0.01-1.77	0.03-15.91	0.01-0.49	0.01-1.35	0.01-0.11	0.01-2.85	0.02-22.11	0.01-1.42	0.05-10.21	0.02-0.45	0.01-0.52	ND	ND
Average	14.4	0.11	0.60	0.03	0.11	<0.01	0.21	1.07	0.04	0.45	0.02	0.04		
Percent Positive sites [1]	100.0	79.6	87.7	40.8	83.7	10.2	89.8	91.8	20.0	34.0	12.0	32.0		
Salt Lake City, Utah														
Range	1.8-74.5	0.01-1.38	0.02-2.64	0.03-0.09	0.01-0.59	0.02-0.17	0.01-0.85	0.01-5.00	0.01-1.14	0.02-7.50	0.01-0.24	0.01-0.23	ND	[2] 0.06
Average	15.7	0.09	0.23	<0.01	0.06	<0.01	0.10	0.49	0.03	0.41	0.01	0.02		<0.01
Percent Positive sites [1]	100.0	62.0	66.0	6.0	64.0	4.0	66.0	72.0	26.0	38.0	12.0	26.0		2.0
Waterbury, Conn.														
Range	0.9-37.9	0.01-1.26	0.01-6.46	0.01-0.67	0.01-1.29	0.01-0.04	0.01-2.27	0.01-10.35	0.02-0.22	0.02-8.74	0.01-0.53	0.01-0.53	ND	ND
Average	8.5	0.14	0.43	0.04	0.13	<0.01	0.19	0.98	0.01	0.96	0.01	0.02		
Percent Positive sites [1]	100.0	44.0	48.0	22.0	50.0	4.0	52.0	56.0	8.0	28.0	8.0	30.0		

NOTE: ND = not detected.

[1] Percent based on number of sites with residues greater than or equal to the sensitivity limits.

[2] One value.

[3] Based on results for 49 sampling sites.

confidence intervals show some distinct variation in levels between the cities, but, in general, there is considerable overlap.

The geometric means of the arsenic values tended to separate the cities into two general classes—those with residues greater than 5 ppm and those with residues less than 2 ppm. Houston and Miami had geometric means of<2 ppm, and the other cities had residues >5 ppm. These variations in elemental arsenic are probably attributable to differences in geological conditions or possibly contamination from industrial or combustion sources rather than to differences in use of various arsenical pesticides.

The residue levels of DDT and some of the other pesticides were too high to have resulted from general environmental contamination, suggesting that they may have resulted from application of pesticides by city governments, home owners, or other urban activities.

In an effort to determine the sources of contamination from DDTR and arsenic in the cities, the soil residue data for each compound were pooled and divided into "lawn" and "unkept" areas. These areas are defined as follows:

Lawn Areas

1. Mowed grass in close proximity to a house, factory, or other structure.
2. Mowed grass in municipal parks or other town-owned or -operated land.
3. Garden or cultivated areas.
4. A yard that is in obvious proximity to a home.

Unkept Areas

1. Vacant lots where grass is apparently uncared for.
2. Small wooded lots, brush, or overgrown fields.
3. Areas such as power lines, gas lines, etc.
4. Bare exposed soil around construction sites, eroded areas, etc.

A "t" test, for unequal numbers of observations, was used to test for significance. When arsenic residues were tested, no significant difference could be detected in residues between lawn and unkept areas. This supports the hypothesis that arsenic residues resulted either from background levels or from general pollution sources such as coal or petroleum combustion. However, a significant difference (t=4.15) was detected for DDTR residues, with the higher residues found in the lawn areas. This indicates that DDTR residues in the city areas probably occurred as a direct result of pesticides used within the city.

In conclusion, the cities sampled in this study generally had heavy loads of chlorinated hydrocarbon pesticide residues in soil. These tended to be greater than cropland residues from the same State, although tests of significance could not be made on these data. There was some distinct variation in residue levels between cities. Finally, within the cities, DDTR residues in lawn areas were significantly greater than those in unkept areas.

TABLE 2.—*Comparison of residue levels of compounds present in both urban and cropland soils in six States*

Location	Residues in PPM			
	Arsenic	DDTR	Dieldrin	Chlordane
California				
Bakersfield	7.1	0.36	0.07	0.78
Cropland	5.2	1.43	0.02	0.01
New Jersey				
Camden	11.2	1.36	<0.01	0.36
Cropland [1]	6.8	0.14	0.02	<0.01
Florida				
Miami	2.3	5.98	0.72	1.59
Cropland	0.8	0.85	0.08	0.36
Wisconsin				
Milwaukee	14.4	1.07	0.04	0.45
Cropland	3.8	0.02	0.01	0.01
Utah				
Salt Lake City	15.7	0.49	0.03	0.41
Cropland [2]	4.8	0.20	0.01	0.02
Connecticut				
Waterbury	8.5	0.98	0.01	0.96
Cropland [3]	10.0	0.59	0.01	0.01

[1] Average of results for New Jersey, Delaware, and Maryland.
[2] Average of results for Arizona, New Mexico, Nevada, and Utah.
[3] Average of results for Maine, New Hampshire, Vermont, Massachusetts, Rhode Island, and Connecticut.

TABLE 3.—*Geometric means and 95% confidence intervals for DDTR and arsenic residues in soil from six cities—1969*

City	DDTR			Arsenic		
	Upper (ppm)	G.M. (ppm)	Lower (ppm)	Upper (ppm)	G.M. (ppm)	Lower (ppm)
Bakersfield, Calif.	0.27	0.20	0.13	6.7	5.5	4.5
Camden, N.J.	0.62	0.37	0.21	10.0	7.6	5.8
Houston, Tex.	0.04	0.02	0.01	1.5	1.1	0.7
Manhattan, Kans.	0.18	0.09	0.04	8.2	6.2	4.7
Miami, Fla.	2.82	1.60	0.91	1.1	0.6	0.3
Milwaukee, Wis.	0.50	0.31	0.19	13.1	8.4	5.4
Salt Lake City, Utah	0.21	0.12	0.06	12.6	9.6	7.4
Waterbury, Conn.	0.18	0.08	0.04	7.8	6.1	4.8

See Appendix for chemical names of compounds discussed in this paper.

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APPENDIX

Chemical Names of Compounds Discussed in This Issue

ALDRIN	Not less than 95% of 1,2,3,4,10,10-hexachloro-1,4,4a,5,8,8a-hexahydro-1,4-*endo-exo*-5,8-dimethanonaphthalene
ARSENIC	As_2O_3
AZINPHOSMETHYL	*0,0*-dimethyl *S*-(4-oxo-1,2,3-benzotriazin-3[4*H*]-ylmethyl phosphorodithioate
BHC	1,2,3,4,5,6-hexachlorocyclohexane, mixed isomers
CARBOPHENOTHION (TRITHION®)	*S*-[(*p*-chlorophenylthio)methyl] *0,0*-diethyl phosphorodithioate
CHLORDANE	1,2,4,5,6,7,8,8-octachloro-3a,4,7,7a-tetrahydro-4,7-methanoindane
γ-CHLORDANE	———————————————————, gamma isomer
DDE	1,1-dichloro-2,2-bis(*p*-chlorophenyl) ethylene
DDT (including its isomers and dehydrochlorination products)	1,1,1-trichloro-2,2-bis(*p*-chlorophenyl)ethane; technical DDT consists of a mixture of the *p,p'*-isomer and the *o,p'*-isomer (in a ratio of about 3 or 4 to 1)
DEF®	*S,S,S*-tributyl phosphorotrithioate
DIAZINON	*0,0*-diethyl *0*-(2-isopropyl-4-methyl-6-pyrimidyl) phosphorothioate
DIELDRIN	Not less than 85% of 1,2,3,4,10,10-hexachloro-6,7-epoxy-1,4,4a,5,6,7,8a-octahydro-1,4-*endo-exo*-5,8-dimethano=naphthalene
ENDRIN	1,2,3,4,10,10-hexachloro-6,7-epoxy-1,4,4a,5,6,7,8,8a-octahydro-1,4-*endo-endo*-5,8-dimethanonaphthalene
EPN	*0*-ethyl *0*-(*p*-nitrophenyl) phenylphosphonothioate
ETHION	*0,0,0',0'*-tetraethyl *S,S'*-methylenebisphorodithioate
HEPTACHLOR	1,4,5,6,7,8,8-heptachloro-3a,4,7,7a-tetrahydro-4,7-methanoindene
HEPTACHLOR EPOXIDE	1,4,5,6,7,8,8-heptachloro-2,3-epoxy-3a,4,7,7a-tetrahydro-4,7-methanoindan
MALATHION	diethyl mercaptosuccinate, S-ester with 0,0-dimethyl phosphorodithioate
MERCURY	Hg
METHYL PARATHION	*0,0*-dimethyl *0*-*p*-nitrophenyl phosphorothioate
PARATHION	*0, 0*-diethyl *0*-*p*-nitrophenyl phosphorothioate
PHORATE	*0,0*-diethyl *S*-(ethylthio)methyl phosphorodithioate
SELENIUM	Se
TDE (DDD) (including its isomers and dehydrocalorination products)	1,1-dichloro-2,2-bis(*p*-chlorophenyl)ethane; technical TDE contains some *o,p'*-isomer also
TOXAPHENE	chlorinated camphene containing 67% to 69% chlorine

Information for Contributors

The PESTICIDES MONITORING JOURNAL welcomes from all sources qualified data and interpretive information which contribute to the understanding and evaluation of pesticides and their residues in relation to man and his environment.

The publication is distributed principally to scientists and technicians associated with pesticide monitoring, research, and other programs concerned with the fate of pesticides following their application. Additional circulation is maintained for persons with related interests, notably those in the agricultural, chemical manufacturing, and food processing industries; medical and public health workers; and conservationists. Authors are responsible for the accuracy and validity of their data and interpretations, including tables, charts, and references. Accuracy, reliability, and limitations of the sampling and analytical methods employed must be clearly demonstrated through the use of appropriate procedures, such as recovery experiments at appropriate levels, confirmatory tests, internal standards, and inter-laboratory checks. The procedure employed should be referenced or outlined in brief form, and crucial points or modifications should be noted. Check or control samples should be employed where possible, and the sensitivity of the method should be given, particularly when very low levels of pesticides are being reported. Specific note should be made regarding correction of data for percent recoveries.

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——Manuscripts should be typed on 8½ x 11 inch paper with generous margins on all sides, and each page should end with a completed paragraph.

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